Essays on the economics of African wildlife utilization and management

by

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Contents

Chapter 1

Introduction and Summary

1. Background

Protected areas have long been recognized as the single most important method of conserving biological diversity worldwide. This practice is more than a century old, but during the recent decades, the areas designated as protected have expanded dramatically. For instance, by 1989 some 4 500 sites covering 4.8 million km^2 (3.2 % of the Earth's land surface) had been protected (Reid and Miller 1989). Among these, 444 sites constituting 0.9 million km^2 were located in the Afrotropical region (Reid and Miller 1989). The International Union for the Conservation of Nature (IUCN) lists eight types of protected areas ranging from strict nature reserves to multiple-use management areas (see e.g. Brown 2000). The areas are ranged by the accessibility for people, where strict nature reserves allow scientific studies only, while the multiple-use management areas allow economic activities like pasture, utilization of timber and wildlife etc. The national park is listed as the second most restricted protected area. One example is the Serengeti National Park in Tanzania which allows scientific studies and non-use tourism activities such as game viewing[1]. This park is also of focus in this thesis. Other examples are the Royal Chitwan National Park in Nepal, the Iguazu National Parks in Argentina and Brazil, and Volcan Poas National Park in Costa Rica (Brown 2000). Today, more than 50 per cent of the protected areas worldwide are in national parks (Brown 2000).

In Africa, wildlife resources and the management of protected areas have received a lot of attention over the past decades. The attention has been stimulated by the observation that protected areas have failed in protecting many wildlife populations on this continent. While the areas designated for protection have expanded, several species are

[1] The Game Reserves on the western border of Serengeti National Park allow sport hunting for tourists and managed cropping of ungulates such as wildebeest, zebra and topi (see chapter 5).

now locally extinct or seriously endangered. The African elephant population declined by 31 per cent (from 1.2 million to 764 000) between 1981 and 1987, mainly due to the demand for ivory (Reid and Miller 1989). In addition, an estimated population of 65 000 black rhinos in 1970 had fallen to 3 800 rhinos by 1986 (Leader-Williams et al. 1990). Today, the rhino population is extinct in Serengeti in Tanzania, whereas elephant poaching has stopped. Scientists claim that the elephant poaching was eliminated in Serengeti due to the world ban on the ivory trade imposed in 1989 (Sinclair and Arcese 1995). Still, Serengeti is continuously experiencing an invasion of poachers, targeting large resident and migratory herbivores, such as wildebeest, impala, and topi (Campbell and Hofer 1995).

Traditionally, the strategy for preventing or restricting human impact and discouraging illegal activities in protected areas has been to establish law enforcement to detect and sentence intruders. Yet, human encroachment has continued to severely degrade and destroy many protected areas. This clearly demonstrates that however important the resources are intrinsically, they are also important because humans still use them. Whether we welcome or deplore this fact, it is still the key to continued wildlife survival. In response to this and the observed failure of protected areas, conservationists and scientists have searched for alternative approaches to wildlife management. This has resulted in the apprehension that successful management must somehow include the co-operation and support of the local people. Excluding people living adjacent to protected areas from use of the resources, without providing them with alternatives and compensation, is today viewed as an unethical and ecologically unsustainable management approach. During the past two decades, projects involving community participation have been launched and implemented. These projects represent a broad set of schemes, ranging from pure benefit-sharing, such as transfers from wildlife related activities like tourism, to a more far-reaching design of community-based management where the local community is trained to manage and control the resources on its own.

This thesis looks at several questions related to the evolvement of wildlife management systems. Three of the papers address these questions within a theoretical bio-economic framework (chapters 2-4), while the final paper offers an empirical analysis of wildlife utilization and management in Serengeti (chapter 5). The first paper asks whether the failure of the traditional 'fences and fines' approach to protected area management is rooted in restricted access to agricultural land. The answer is partly yes; if agricultural land is expropriated for protection, the local people may compensate their loss by means of a more extensive use of the wildlife. This result motivates the second question: is it possible to promote wildlife conservation by handing the property rights to wildlife over to the local people? The answer to this question is unclear. It turns out that local management may cause wildlife degradation in areas with severe costs of living with wildlife (i.e. extensive wildlife-induced damage in agriculture). Third, can a management system based on benefit sharing, which lies somewhere between the previous systems, be the solution to wildlife protection? The answer is yes if the benefit-sharing scheme is properly related to the conservation objective. The empirical analysis of Serengeti shows that the benefit-sharing project established in this area has reduced the hunting pressure. In addition, policies which encourage the local people to sell agricultural crops on the market may promote wildlife conservation.

2. Property rights

2.1. Property, property rights, and property rights regimes

Although most economists agree that institutions and property rights are important determinants of natural resource exploitation, the focus on this has been both limited and confusing in standard economic texts. In general, the functioning of the property structure can be used to classify three types of property rights regimes. First, we have state or private property, where there is a sole owner of the resource, i.e. the State or a private agency. The national park is one example of this regime. Analyses of state or private property are based on efficiency; the concept of rent maximization where the sole owner treats the resource as an asset. Then, the value of uncaught resources, i.e. the shadow price of the resource, prevents the owner from over-exploitation as long as he expects to be the one to benefit from the current 'conservative' harvesting policy, or investment, in the future. Technically, having a sole owner is in accordance with the classical Clark (1973) model[2].

Second, let us consider the open-access regime. The theory of open-access exploitation was first developed by Gordon (1954) and is explained follows. A particular resource can be regarded as open-access property if there is no possibility of excluding harvesters attracted by excess rents. In this regime there is no legally defined ownership and every agent is free to exploit the resource. Then, no harvester can be sure of who will benefit from the value of uncaught resources. New agents will enter the industry, or existing harvesters will expand their effort use, as long as there is positive profit to gain. In this way, open-access gives entry until zero resource rent. Therefore, all else equal, open-access leads to economic over-exploitation compared to sole ownership.

It is important to note that the term economic over-exploitation is not equivalent with what we understand as ecological over-exploitation. Economic over-exploitation occurs when the industry fails in maximizing the economic rent. However, biologists consider a resource to be over-exploited only when the population has been reduced below the level of maximum sustainable biological yield, i.e. below the level generating the maximum natural growth of the resource (see e.g. Clark 1990). Whether the open-access regime results in ecological over-exploitation is strictly dependent on the economic and ecological conditions in place. The open-access extraction level resulting from a 'high' cost-price ratio is likely to be below the level which generates the maximum sustainable biological yield of the resource. Then, open-access gives economic over-exploitation but no ecological over-exploitation. On the other hand, a 'low' cost-price ratio may give both economic and ecological over-exploitation in the open-access regime.

Somewhere between sole ownership and open-access lies the common property rights regime, in which a small number of private agents constitute a group of co-owners. In

[2] However, Clark (1973) shows that present-value maximization may cause extinction of the resource if the discount rate sufficiently exceeds the maximum reproductive potential of the population. See also Plourde (1970) and section 4.1 below.

early economic research on property rights, common property was understood as open-access (see Hardin 1968). In this way, the literature overlooked the possibility that conformity and co-operation within the group of owners may result in a well-functioning common property regime. That is, through co-operation the group of owners may succeed in totally internalising stock externalities and maximizing rent. There have been a number of studies trying to understand the necessary conditions for a well-functioning common property regime. It has been pointed out that co-operation is more likely to succeed in small homogenous groups where prevalence of individual conformity to group norms is likely to be valid, while group heterogeneity and conflicting interests may impede co-operative agreements. See e.g. Bardhan (1993) and Seabright (1993). There are several documented examples of co-operation among owners, also on the local community level in developing countries. One contributor is Ostrom (1990), who cites evidence from several parts of the world.

Today, there is a considerable interest in the literature to solve the confusion between common property and open-access. In order to do this, Bromley (1991) emphasises that property is a benefit stream rather than a physical object (i.e. wildlife stock, fish stock, forested area etc.). He claims that it is the traditional understanding of 'property' as a physical object which has caused the confusion around the common property and the open-access regimes: when interpreting 'property' as the resource stock, and if resource exploitation is available to all who might be interested, then the 'property' is thought to be 'commonly available to all'. Instead, we should think of 'property' as a benefit stream and a property right as 'a claim to a benefit stream which the state will agree to protect through the assignment of duty to others who may covet, or somehow interfere with, the benefit stream' (page 2). To have a property right to something means to have a protected right to the benefit stream arising from that situation. The effective protection which the right holder(s) gains from the State is a correlated duty for all others interested in his (their) claim. Thus, Bromley (1991) interprets property as a triadic social relation involving a benefit stream, the right holder(s) and the duty bearers. Rights are not relationships between the right holder(s) and the object, but are rather relationships between the right holder(s) and the others with respect to that object. Further, a right exists only when the State agrees to defend a right holder's interest by ensuring that those with duty respect their interests. So, possession of inviolable property rights presupposes that the rights are, firstly, authorised by law and, secondly, effectively enforced and protected by the State.

The interpretation of property as a triadic social relation helps us to distinguish between the common property regime and open-access without the confusion discussed above. The common property regime represents a case of legally well-defined ownership in which the co-owners have a right to reap the benefits of the resource while the others have a duty not to interfere with their interest. Under appropriate designed institutional arrangements, these rights are perfectly protected by the State. In contrast, there is no legally defined ownership in the open-access regime and no one has the legal right to keep any potential user out. The resource is therefore available to the first party to effect capture.

As stated above, a well-functioning common property regime requires the presence of a social mechanism to ensure co-operation within the group of co-owners. Even if the co-owners should fail to co-operate, outsiders will still have a duty to respect the owners' rights. However, within the group of owners, the common property regime degenerates into an open access regime with economic over-exploitation of the resource.

2.2. The property rights regimes of the thesis

In this thesis, all papers are centred on wildlife conservation and utilization in and around protected areas. The category of protected areas in mind is the national park. The property rights of most national parks in the developing world are well defined and, usually, in the hands of the State (see Brown 2000). In this setting, the State has the legal rights to manage and control the use of the resource. One way to do this is to impose anti-poaching laws and enforcement in order to restrict the impact of non-owners who claim their rights through illegal utilization. This is the point of departure in chapter 3 to 5, where the local people utilize the wildlife stock through illegal hunting. A second alternative for the State is to lease the resource to groups or individuals who are thus given user rights for a specified period of time (see Bromley

1991). This is the case in chapter 2, where the local people are given legal rights to wildlife hunting. A third alternative is to alter the property rights structure by handing the property rights over to the local people, i.e. implement a common property regime. In this case, the State fully transfers the rights to manage and control the resource over to the local community. The effect of altering the property rights regime is investigated in chapter 3.

In this section it is of interest to explain how the thesis intuitively and technically distinguishes the regime of no property rights to the local people (i.e. state property) from the regime where the local people hold the property rights (i.e. common property). The important aspect here is whether the local people are able to fully control the resource stock through hunting, an ability which is strictly dependent on the property rights regime in place. All analyses assume that the local people are able to co-ordinate the hunting activity. I am looking away from conflicting interests among them and, hence, prevalence of individual conformity to group norms is assumed to be present[3]. Therefore, possible shifts in the capability of the local people to control the stock do not result from any change in the social relations within this group. Instead, such shifts are caused by a changing property rights regime.

Let us first consider the case where the property rights are in the hands of the State. In this scenario the State has the legal rights to manage and control the resource. Then, although the local people are able to co-ordinate the hunting, they are unable to control the resource stock through this activity. This follows from the property rights regime, where the lack of property rights for the local people restricts their hunting decision. In the essays of the thesis the reason the state property regime imposes such a restriction is due to the respective institutional settings. In chapter 2, the local people are given user rights to wildlife, while the property rights are in the hands of the State. The extent of the user rights is determined by the State, which has the authority to restrict these rights by reducing the size of the legal hunting grounds. In fact, the local people experience a continuous risk of being denied future access to these grounds. The uncertainty related

[3] In line with traditional reasoning it is assumed that the elders are in charge of the group's activities (Marks 1984).

to their future rights restricts their hunting decision and unable them to fully control the stock. In chapters 3-5, however, it is assumed that the local people have no legal user rights and, hence, hunting is illegal. Here, the presence of anti-poaching law enforcement, such as guards and patrol units, is what restricts their hunting activity and makes it difficult for them to fully control the resource stock. Technically, the lack of control means that the local people are not taking into account the stock effect of their harvesting decision. Hence, they treat the resource stock as *exogenous* when deciding their harvesting activity. This approach is also used by Smith (1969, 1975).

The scenario is quite different when the local people have the property rights to the wildlife. In this case, it is assumed that their rights are effectively enforced by the State, meaning that all others are excluded from the use of the resource. As long as the local people's rights are effectively protected by the State, they expect that they are the ones to control the benefit stream in the future and, consequently, they will take into account the stock effect of their harvesting decision. The group of owners will therefore behave as the sole owner in a private or state property regime and succeed in internalising stock externalities and maximizing rents (see section 2.1). This means that the local people are able to fully control the resource through the hunting activity.

Technically, this regime may be interpreted in two ways. First, in an analysis of sole ownership, which is similar to the well-functioning common property regime, Smith (1969) assumes that the owner maximizes net present revenue when taking the stock effect into account. Hence, he presents a static model where the resource owner considers the stock as *endogenous* when maximizing net revenues. However, Clark (1973) claims that it may be more reasonable to assume a dynamic model where the resource owner takes into account the preference for present over future revenues. The second approach is therefore to consider the resource as a capital stock in that it is capable of yielding a sustainable harvest flow through time (see also Clark and Munro 1975). As in capital theory, the present harvesting decision will have implications for the future harvesting options through its impact on the stock level. The problem of the resource owner thus becomes one of selecting an optimal harvesting flow through time. The resource owner maximizes the *present value* of the harvest sequence by discounting future revenues at some fixed rate. Then, a higher discount rate increases the value of harvesting, or disinvestment, relative to the 'own rate of return' from investments in the resource stock. Hence, in contrast to Smith's static model (1969), a positive discount rate implies that it is not optimal to invest in the resource to the extent that the current resource rent is maximized (see also Plourde 1970). However, when the rate of discount equals zero, the dynamic model coincides with the static model (see e.g. Munro and Scott 1985) [4].

In this thesis, the case of a well-functioning common property regime is considered a special case of the present-value maximization, in which a zero discount rate applies. Hence, this regime is implemented in its most optimistic way in the sense that disinvestments in the resource today is valued at its lowest possible level. What distinguishes this regime from the regime of no property rights to the local people is therefore that the local people consider the wildlife stock as endogenous, i.e. they take the stock effect into account when determining the optimal harvesting strategy. A more detailed discussion of the institutional settings of this thesis is given in the respective chapters.

3. Protected area management; designs and possible pitfalls

Despite the extensive network of protected areas and imposed anti-poaching laws, poaching has been a continuous threat to wildlife in Africa (see section 1). Today, there is a growing understanding among conservationists and scientists that the traditional approach of relying solely on law enforcement in the protected areas, has failed to protect wildlife on this continent. The exclusion of the local people, usually without compensation for the loss of property, title and traditional hunting rights, has worked against their economic interests and led to increased antagonism and a disincentive to conserve wildlife (see Marks 1984, Kiss 1990, Brandon and Wells 1992, Milner-Gulland and Leader-Williams 1992b, Swanson and Barbier 1992, Wells 1992, Nepal and Weber 1995). A rethinking of wildlife management schemes has emerged, where

[4] Note that the open-access regime with zero resource rent can be considered as a special case of the present-value maximization, in which an infinite discount rate applies.

the main strategy is to somehow include the local people to gain their co-operation and support. By stimulating local support, the conservation managers aim at changing the current pattern of exploitation and limiting or preventing uses that endanger wildlife and other natural resources.

In some instances, the management authorities have been advised to de-gazette protected land. For example, Malawi suffers from extreme rural poverty and increasing scarcity of arable land, yet 11 per cent of the country's land area is set aside as protected areas. In comparison, the world average is 4 per cent (Swanson and Barbier 1992). It has been argued that some of the protected land in Malawi could safely be de-gazetted and redistributed to the local people for agricultural use (Swanson and Barbier 1992). Clearly, however, there will always be a limit as to how much protected land can be de-gazetted without seriously affecting wildlife and its resource base.

Possibly a more widely applicable approach, and one which has been promoted vigorously by international organizations since the 1980s, is to reconcile local support through Integrated Conservation and Development Projects (ICDP). These projects aim at enhancing conservation through approaches that address the needs, constraints and opportunities of the local people (see Kiss 1990, Barbier 1992, Brandon and Wells 1992, Wells and Brandon 1992, Barrett and Arcese 1995, Songorwa 1999). Local participation is launched as the means to improve the economic conditions of the local people, and thus discourage poaching and promote wildlife conservation. Well-known ICDPs in Africa are the CAMPFIRE and WINDFALL in Zimbabwe and the ADMADE in Zambia (see Kiss 1990, Wells and Brandon 1992, Barrett and Arcese 1995, Gibson and Marks 1995, Gibson 1999). The Kenya Wildlife Service has adopted a benefit-sharing strategy where revenues from protected areas are distributed to local communities through a Wildlife Development Fund (Emerton 1998). Another example is the Serengeti Regional Conservation Project (SRCP) in Tanzania which is of focus in chapters 4 and 5 of this thesis (see also Barrett and Arcese 1998).

In a review of existing ICDPs in Africa, the World Bank discusses three levels of local participation to promote the joint goal of wildlife conservation and rural development

(Kiss, 1990). These are (1) participation through benefits and compensation; (2) participation in planning and design; and (3) participation in implementation and management. When addressing the challenges of ICDPs, this thesis centres on projects based on strategies (1) and (3). The next step is to define and describe these levels of local participation and reveal some problems related to their design. First, participation through benefits and compensation, i.e. level (1), represents a direct way of improving local income, and is today a generally accepted principle in wildlife conservation programmes. Usually, the local communities benefit through money transfers from the wildlife tourism sector, village job creation in tourism and park services, and user rights or substitutes for resources in which access has been denied. For the latter, game culling – i.e. distribution of game meat from managed harvests – is implemented in several existing projects. For projects based on this participation level, the property rights to wildlife are in the hands of the State or the management authority, while the local people have no legal property rights. Second, local participation of level (3) is more far-reaching than level (1) in that it empowers the local communities to manage the wildlife on their own. In its broadest extent, this level of participation is founded on common property management, where the local community has the legal property rights to manage and control the wildlife. Hence, participation levels (1) and (3) represent different property rights regimes.

While the concept of ICDP is viewed as a promising method of conserving wildlife in protected areas, experience so far reveals several design dilemmas. Brandon and Wells (1992) give a broad discussion of such dilemmas and highlight possible pitfalls regarding both levels of local participation described above. First, experience from projects based on participation through benefits and compensation shows that many existing schemes lack a clear link between the benefit/compensation strategies and the conservation objective. As pointed out by Kiss (1990) and Brandon and Wells (1992), without such a link it is difficult for the local people to understand that there is a purpose of improved conservation behind the benefits they receive. If worst comes to worst, they may regard the benefits as lump-sum transfers and carry on the exploitation activities as before. This issue will be discussed further in chapter 4.

Second, the concept of community wildlife management also faces some difficulties. As discussed in section 2.1, heterogeneity and conflicting interests among the community members may threaten the performance of the community as a resource manager. Another important issue is the one of conflicting interests between the project manager, i.e. the State, and the local people (Brandon and Wells 1992). While the project defines decline in species as the major problem, this may not be of concern for the local community. For instance, in areas where wildlife cause extensive damage to agricultural crops and livestock, the local people are likely to be more concerned with preventing such damage by getting rid of 'problem' animals than conserving the wildlife stock. If this is the case, giving the local community full control of the management may be a disaster for the wildlife. This issue is addressed in chapter 3 of this thesis.

4. The bio-economic theory

4.1. Bio-economics and resource extinction

Bio-economic modelling – i.e. modelling which combines ecological systems and economic conditions – is the analytical tool applied throughout this thesis. The tool is powerful because it explicitly integrates the influence of human economic activity on environmental conditions. In natural sciences, several attempts have been made to assess the magnitude of illegal activities in and around protected areas, like quantifying the number of exploiters and the damage imposed on the natural resources in these areas (for the case of Serengeti, see Sinclair and Arcese 1995). However, less attention has been paid to the underlying motivation for exploitation, which is of considerable importance to the management of protected areas. Bio-economic modelling extends this literature by focusing on the incentives to exploit natural resources. In doing so, it addresses important issues and gives valuable policy implications for the management of natural resources.

The early focus of bio-economic models, represented by Gordon (1954) and Clark (1976), was on marine resources. As mentioned in section 2.1, this literature examined the consequences of open access and demonstrated that this regime leads to economic over-exploitation. Further, it was shown that the degree of exploitation increases with the price-cost ratio. More recently, Mesterton-Gibbons (1993) has analysed resource utilization and property rights regimes in a game-theoretic framework. He demonstrates that in a well-functioning common property regime the resulting stock level is independent of the number of owners. The reason for this is that the group of co-operating exploiters will behave as a sole owner. The aggregate offtake is therefore independent of the number of exploiters, while the offtake of each individual exploiter is reduced with this number. The latter result occurs because the additional exploiter imposes additional costs on existing owners through the stock externalities. On the other hand, in a non-cooperative game the stock level shrinks when the number of exploiters increases, and the system approaches the open-access solution with zero rent as this number approaches infinity.

While this literature links resource depletion to the property rights structure, Clark (1973) demonstrates that resource extinction might appear as the most attractive policy even for the sole owner. He analyses extinction as a process of human choice between which productive assets to retain in the natural portfolio (see also Clark and Munro 1975 and section 2.2). The sole owner maximizes the present value of rent derived from resource exploitation. The optimal strategy is to invest in the resource until the marginal return from that asset equals the marginal return from other assets. Clark finds that extinction is optimal if the harvesting price exceeds the unit cost and the discount rate is sufficiently large. The policy implications of the early bio-economic models are therefore straightforward: within both poorly managed access and sole ownership, the probability of extinction can be reduced through policies which cause the price-cost ratio to decline.

Because the early models of resource exploitation were developed within the literature of fishery economics, there were assumed to be no alternative uses for the habitat of the resource. Competing uses for the marine habitat (i.e. the sea) were not of interest and the opportunity cost of using the sea for fish production was assumed to be zero. However, on land there clearly exist alternative uses for the habitat of given species, such as converting land to agricultural crop production or as grazing land for livestock, both of which represent important alternative uses in rural areas in developing countries. Swanson (1994) has extended the bio-economic framework to jointly analyse the habitat (i.e. land) and its species. He assumes that the natural growth rate of a biological resource is affected by the allotment of the natural habitat. Within this framework he demonstrates that species will receive allocations of habitat only to the extent where the species are able to generate competitive rate of return from this use of land. Hence, a non-competitive resource is subject to more than one form of disinvestment: in addition to direct harvesting, resource depletion results from land being converted to an alternative use that yields a higher return.

Since the initial modelling of resource extinction the approach to the problem has altered from well-focused concerns about the endangerment of individual species to a much broader concern which includes several interdependent species and whole ecosystems. Modelling the natural resource in the means of the dynamics of one single independent population implies a neglect of ecological as well as economic interactions. A single industry (or agent) that exploits several species may severely affect the dynamics and stability of the whole corresponding ecosystem. On the other hand, if several agents exploit different but biologically interrelated species, the exploitation will typically be hampered with externalities between the agents (see Clark 1990).

As pointed out by Clark (1990, chapter 10), multispecies systems are more difficult both theoretically and practically than problems involving one single species. In the case of open-access he demonstrates that extinction cannot occur in a single species Gordon-Schaefer model because, as the stock approaches zero, the unit harvesting cost eventually exceeds the price. However, when studying the joint harvesting of two ecologically independent species, he shows that one population may be driven to extinction. In addition, Clark (1990) demonstrates different systems of interacting populations: the predator-prey relationship in which the predator feeds on the prey and the Gause model where the species compete over a common grazing area.

4.2. The present bio-economic literature on wildlife utilization and management

The following presents a review of the recent bio-economic literature on wildlife harvesting and management. This literature comprises several types of models, each of which captures important economic aspects of wildlife conservation relevant for the analyses presented in this thesis. First, there is a class of bio-economic models focusing on illegal wildlife hunting and anti-poaching law enforcement. In these models, the resource owner invests in enforcement to protect certain species and the legal benefits they provide. Typically, the resource 'owner' is the manager of a protected area. These models do not take competing land uses into account, as recognized by Swanson (1994) (see section 4.1), but focus solely on anti-poaching law enforcement, hunting effort and the offtake of wildlife. Such a model has been explored by Milner-Gulland and Leader-Williams (1992a), who investigate how law enforcement, in relation to changes in the detection rate and the penalty level, affects the poachers' decision to hunt black rhinos and elephants in Luangwa Valley in Zambia. They distinguish between the hunting incentives of local subsistence poachers and professional gangs involved in commercial hunting. Subsistence hunting is defined as hunting performed by the local people for the purpose of domestic consumption or to sell wildlife products at the local market. On the other hand, commercial hunting is carried out by professional gangs from outside the local communities. These gangs sell the wildlife products further afield, often at the international market place (Milner-Gulland and Leader-Williams 1992b). In the case of a Schaefer production function, Milner-Gulland and Leader-Williams (1992a) demonstrate that it is not worthwhile for the local poachers to hunt any of the two species, while it is worth hunting elephants for the professional gang. In the same setting, they show that a penalty which varies with a poacher's offtake is a more effective tool against poaching than a fixed penalty.

Skonhoft and Solstad (1998b) are other contributors to this line of research. They focus solely on a group of subsistence hunters, and demonstrate that lack of law enforcement may result in a wildlife stock as abundant as when law enforcement is present. In addition, Bulte and van Kooten (1999) analyse the effect of anti-poaching enforcement and the ivory trade ban on illegal exploitation of elephants. The effect of implementing a trade ban is generally unclear. However, a sensitivity analysis from Zambia indicates that the ivory trade ban is more effective in conserving elephants than permitting open trade.

Second, there is a class of models analysing protected areas, explicit or implicit, through different land uses. A group of these models consider a single agent (the State, a private agent or a group of local people) who owns land and performs two conflicting land uses; land as a wildlife habitat and land as input in agricultural production. Then, conversion of land for agricultural production displaces wildlife and vice versa. Brought about by conflicting land uses, the single agent models demonstrate how disinvestments in wildlife may occur, despite sole ownership to the resource. In the same way as demonstrated by Swanson (1994), Skonhoft (1999) shows that increased profitability in agriculture is a threat to wildlife because it triggers land use conversion. See section 4.1. Skonhoft and Solstad (1998a) study an implicit land use conflict as a group of livestock keepers faces a trade-off between keeping wildlife and livestock as assets, where the two populations compete in grazing. The owners determine the optimal investment in wildlife by balancing the benefits from wildlife and livestock. In this setting, they demonstrate that low prices for wildlife products may represent a threat to wildlife. See also Skonhoft (1998).

Schulz and Skonhoft (1996) focus on conflicting land uses in a model with two agents. First, there is an agency (the state or private) which has the property rights to wildlife and practices protected area management as a land use. This agent benefits from wildlife culling and tourism. The second agent consists of a group of local people who utilize the land surrounding the park in agricultural production. In this model, no illegal hunting takes place. The characteristics of a social optimum are quite similar to the single-agent model of Swanson (1994). However, Schulz and Skonhoft (1996) consider another important issue in their model, namely that wildlife induce damage to agricultural crops (see also Huffaker et al. 1992, Carlson and Wetzstein 1993, Bulte and van Kooten 1996, Zivin et al. 2000). By implementing crop damage, they demonstrate that a competitive solution may result in a wildlife stock above that of the social optimum. A similar correlation is shown in chapter 3 of this thesis. Chapter 2 is also a contribution to this line of research and adds some new and important insight into land allocation and wildlife conservation.

The final class of bio-economic models of consideration here consists of models which analyse the impact on wildlife conservation of various benefit-sharing schemes, i.e. local participation of level (1) in section 3. Barrett and Arcese (1998) present a household model in order to investigate the effect of game meat distribution from managed harvests to the local people. The local people perform illegal subsistence hunting and Barrett and Arcese find that game meat distribution succeeds in discouraging poaching. However, because the total of the illegal and legal offtake increases, this policy leads to wildlife degradation. Skonhoft (1998) analyses a similar benefit-sharing scheme but reaches the opposite conclusion regarding wildlife conservation. In principle, what makes these results differ is that Barrett and Arcese consider the local people as the active agent, while the active agent in Skonhoft's model is the park agency which earns income from legal hunting and tourism. By forcing the park manager to transfer a fixed share of the wildlife harvest to the local people, the return from hunting is reduced relative to the return on wildlife in tourism. Hence, the park manager responds by making further investments in wildlife.

One important feature of Barrett and Arcese's model is that it is a household model, which opens up the possibility of investigating the impact of failures in the markets for agricultural output and game meat. They assume that no market exists for game meat, resulting in the individual household consuming the whole amount of the illegal offtake domestically. The decision problem of the local people is then a problem of utility maximization where game meat distribution affects the decision problem by altering the budget constraint. On the other hand, if markets exist for both agricultural output and game meat, the decision problem of the local people is similar to the problem of a profit maximizing firm producing two types of products. In this case, game meat distribution works as a lump-sum transfer without any effect on the hunting decision of the local people. Chapter 4 of this thesis analyses benefit sharing in a household model where the group of the local people is the active agent. This model combines the first and the third groups of models above and demonstrates, in contrast to Barrett and Arcese (1998), that distribution of game meat may promote wildlife conservation if properly linked to the costs of being caught in illegal hunting.

The bio-economic literature consists of a broad range of theoretical models analysing wildlife utilization and management. However, less emphasis has been put on empirical analyses of the human-wildlife interface, probably due to the lack of data. Some efforts have been made in natural sciences. For instance, Campbell and Hofer (1995) and Hofer et al. (2000) have estimated the costs and benefits of illegal hunting in Serengeti using records from law enforcement patrols. Campbell and Hofer (1995) estimate the 'profitability' of hunting based on an estimated probability of being caught and the wildlife density in each sub-area (grid cell) of Serengeti National Park (SNP). They find that hunting is 'most rewarding' in the western sector of SNP. This is also the finding of Hofer et al. (2000), who add some information about the offtake value, opportunity cost of hunting and cost related to confiscated weapons. In Lunagwa Valley in Zambia, records from patrols indicate that patrolling affects the level of illegal hunting as poachers tend to be seen less often in more heavily patrolled areas (see Leader-Williams et al. 1990). However, no one I am aware of has estimated functions of hunting effort for the purpose of exploring the incentives behind wildlife exploitation. Therefore, based on household data from western Serengeti, chapter 5 of this thesis gives an empirical analysis of the incentives to hunt.

5. Human behaviour and economic theory

The basic assumptions about behaviour in mainstream neo-classical economic theory are self-interest and rationality. People are assumed to want to get as much as possible out of the feasible alternative actions, and to be able to take into account whatever information readily available to assess the consequences of each alternative and choose the best one[5]. Although these assumptions have made economics successful as a discipline, they have also led to a belief that economic methodology is inadequate for understanding important aspects of human behaviour, particularly in settings in which

[5] See e.g. Gravelle and Rees 1992 for a more accurate definition.

individuals are concerned about the opinions, opportunities and well being of others. There are several examples of social aspects reflecting these concerns, such as culture, co-operation, altruism, laws and social norms. Other social sciences have repeatedly questioned how economic theory can be used to explain social relationships like these (see Elster 1989). The concern about the apparent exclusion of social motivations and arrangements leads some to even question whether the traditional analysis of standard economic problems is coherent (see Akerlof, 1984).

Looking back on the confusion around open-access and common property as discussed in section 2.1, the misconstruction results from the use of very simplistic behavioural and institutional assumptions where, among others, people are assumed to be driven by self-interests only and unaffected by social aspects and the particular institutional arrangements in place. However, empirical and experimental studies suggest that integrated systems of social norms and co-operation are crucial for individual motivation and behaviour in local resource management (see Ostrom 1990). The well-functioning common property regime is more than an accidental collection of independent individuals: it is a 'small' group of people in which the individual members relate to each other according to specific conventions on co-operation and co-existence.

Especially within poor rural communities, some social forces may originate from the need to establish a mutual insurance against draught, pests and so forth in agriculture. Other social norms may be founded on religion, cultural traditions etc. There is a lot of debate on whether adoption to social norms etc., especially when there is no system of law to enforce them, is consistent with the conception of rational individual behaviour (see e.g. Elster 1989). Dasgupta (1993, chapter 8.6) claims that individuals accept social norms and conform to them out of simple self-interest: if a person does not conform, he will suffer from sanctions and it may therefore be rational to stick to the norm[6]. Analytically, there are two ways of addressing the issue of social forces in economic models. Firstly, social forces may be seen as a binding constraint limiting the choices of

[6] It is beyond the scope of this chapter to discuss the economic explanation of norm conformity and co-operation. For a broad discussion, see Dasgupta (1993) and Elster (1989).

a maximizing self-interested individual, and secondly, they may be integrated in the preference structure of the individual agents (Baland and Platteau 1996, page 116).

There are other aspects of human behaviour besides social forces which are important to recognize in economic modelling. In traditional economic models, behaviour is driven by utility or profit maximization where consumers' utility and producers' profit depend on a rather limited set of arguments. However, the failure to recognise alternative objectives for resource exploitation may cause miscalculations on the level of resource use and unsuccessful policy recommendations. For instance, several contributors have argued that the motivation for livestock keeping in poor rural developing societies differs from the assumption of profit maximization. Doran et al. (1979), Smith (1992), Perrings (1993) and Dasgupta and Mäler (1995) claim that livestock keeping is influenced by the fact that the animals are regarded as an important store of wealth as well as income. People in poor rural societies are often constrained in their access to credit, insurance and capital markets, and, consequently, farmers accumulate wealth by keeping extra livestock. In addition, the animals are prone to dying when rainfall is scarce which leads the farmers to keep more animals as an insurance against drought. Moreover, the numbers of animals are more important than the value as far as prestige and status within the community are concerned. While the standard theory will suggest that livestock keeping is motivated by profit maximization, motives based on insurance, prestige and status may cause the *number of livestock* to be more important than the *income* from livestock. This observation does not weaken the assumption of rational individual behaviour, but instead it points out the need to adopt the proper individual objective function.

It is beyond the scope of this thesis to give a broader discussion of the concept of self-interest and rationality. Instead, when analysing a co-operating community it is simply assumed that there is present an underlying integrated system of social norms which encourages people to co-operate. Hence, in accordance with the discussion of common property in section 2, this ensures that the community acts as a single decision-maker. When it comes to the objective function of the decision-maker, it is assumed that behaviour is driven by utility and profit maximisation. However, the reader should keep the above in mind when reading this thesis.

6. An overview of the thesis

The present thesis addresses the performance of wildlife management systems in Africa in general and in the Serengeti-Mara ecosystem in Tanzania in particular. The essays to be presented were initially motivated by the failure of the traditional management of protected areas and the observed difficulties regarding the design of wildlife management systems based on local participation. The thesis provides, based on economic incentives, an explanation of the failure of the traditional protected area management. In addition, the observed broad range of management systems calls for studies of different property rights regimes. In addition, founded on both theoretical and empirical analyses, three essays discuss under what conditions programmes based on local participation will succeed in promoting wildlife conservation.

The methodology of the papers is based on bio-economic modelling and represents all classes of models reviewed in section 4.2. The starting point of every model is a protected area surrounded by a local community. The property rights to wildlife are legally well defined and held by the State or the park manager appointed by the State, i.e. a state property regime. The Sate offers the local people user rights to wildlife in chapter 2, while, hunting performed by the local people in absence of property rights is considered illegal in chapter 3 to 5[7].

The main contribution of the papers is to explore the incentives of the local people to exploit wildlife, and how these are affected by different policies. Throughout the papers the local people are considered the only active agent and, hence, the park manager is

[7] In reality, other aspects such as corruption may be a threat to the performance of the state property regime. For instance, in some countries poaching is successful because government and park officials enter into agreements with the poachers where the poachers share the profit with these agencies in return for protection from being arrested or convicted (see e.g. Gibson 1999, Brown 2000). Throughout this thesis I do not take into account any corruption among the government and park officials.

passive. This means that the implemented policies are treated as exogenous in the models. The effect of the various management strategies is therefore analysed in the framework of partial equilibrium models.

Throughout the thesis, the hunting activity of the local people is interpreted as subsistence hunting (see section 4). According to the kind of problem analysed, the models vary in their specification of both the ecological and economic part. In the ecological part, chapters 2 and 3 provide explicit specifications of two distinct patterns of wildlife migration, while this is not of focus in chapters 4 and 5. In the economic part, chapters 2 and 3 consider the local people as producers of game meat (through wildlife hunting) and agricultural output, with a market for both outputs. Then, the decision problem of the local people can be compared to that of a firm producing two types of products. In contrast, chapters 4 and 5 regard the local people as a group of households who jointly determine their production and consumption decisions. The models presented here allow for market failure and domestic consumption of all quantity of a particular output. A closer overview of the essays follows below.

6.1. Wildlife conservation, human welfare and the failure of protected Areas

The traditional approach to wildlife management has been to gazette land for wildlife protection without offering the local people any compensation or alternatives for the restricted access to land and wildlife. Chapter 2 investigates the impact of this policy on wildlife conservation and human welfare. The starting point is a park agency appointed by the State. The park agency holds the property rights to wildlife inside as well as outside the protected area. Hunting is illegal in the protected area and law enforcement is efficient, meaning that the property rights of the park authority are efficiently protected and illegal hunting does not take place. However, the local people are given user rights to wildlife roaming in the outer area.

The ecological model presented in this chapter is identical to the one developed by Hannesson (1998). Here, wildlife migrates between the protected area and the outer area in a density-dependent way. Species therefore flow from the densely populated area to the area with a low wildlife concentration. In the economic part of the model the local people living in the outer area are involved in two production activities: agricultural production and legal wildlife hunting. Hunting is not the only component of the human-wildlife interface because, when outside the protected area, wildlife destroys agricultural crops. Living with wildlife is therefore twofold: it represents a benefit due to hunting and a cost due to agricultural damage.

This analysis draws on existing bio-economic literature on marine reserves where, Conrad (1998), Hannesson (1998), Pezzey et al. (2000), and Sanchirico and Wilen (2001) demonstrate that marine reserve creation in an open-access fishery increases the aggregate fish stock. The fishery is defined as better off if the reserve increases the total harvest. Because the fish stock disperses between the reserve and the open patch, the effect on total harvest of closing one patch is positive if increased dispersal from the reserve compensates for the foregone harvest in the reserve. In this way, the literature demonstrates that there is a potential for a double payoff of marine reserve creation, where both the fish stock and total harvest increase.

This paper adds to the research of marine reserves as it embraces the theory that there may be an alternative use for protected land as an input in agricultural production. It is assumed that the local people living in the outer area cultivate land for crop production. The performance of protected areas is therefore strictly dependent on the *type* of land gazetted. One alternative is to establish a protected area by gazetting non-cultivated land. This strategy is quite similar to marine reserve creation, and I show that it promotes wildlife conservation. However, the impact on human welfare is negative. The second way to establish a protected area is through expropriation of cultivated land. An alternative cost of habitat protection is therefore present, and it equals the foregone return from agricultural production. I demonstrate, in contrast to marine reserves, that this policy may cause wildlife degradation and poorer economic conditions for the local people. Hence, this chapter gives a bio-economic explanation of the failure of protected areas. Analysing the driving forces behind such an outcome is the main contribution of this paper.

6.2. Property rights and natural resource conservation. A bio-economic model with numerical illustrations from the Serengeti-Mara ecosystem

In chapter 3, which is co-authored with Anders Skonhoft, we examine how wildlife conservation is affected by altering the property rights to wildlife in the outer area. The ecological model presented here differs from the one in chapter 2 as the wildlife migration is seasonal and not density-dependent. Wildlife migration is related to food supply, an ecological specification that fits the migration of wild herbivores in the Serengeti-Mara ecosystem. The resulting ecological system is quite similar to the model of Homans and Wilen (1997).

The state has the property rights to wildlife both inside and outside the protected area. Law enforcement is effectively preventing illegal hunting in the protected area, while illegal hunting takes place in the outer area due to lack of enforcement. As in chapter 2, the local people living in the outer area are involved in wildlife hunting and agricultural production. Here, however, the land use in agriculture is fixed and we only focus on how property rights affect the distribution of labour effort between these activities. Two property rights scenarios are considered. First, the local people have no property rights to wildlife and hunting in the outer area is illegal. This is in accordance with the stylised facts in Africa today. In the second scenario, the property rights to wildlife roaming in the outer area are handed over to the local people. Because the local people are considered a homogenous group of co-owners, this case captures the impact on wildlife conservation of a well-functioning common property rights regime.

While local management is viewed as a superior aim in the literature of community-based wildlife management, this analysis shows that local management may fail in conserving wildlife. This means that even if the local people co-operate, there may be conflicting interests between them and the project manager (i.e. the State) responsible for the implementation of local property rights. The project manager presumes that property rights encourage the local people to invest in wildlife. However, with extensive costs of living with wildlife, the local people will increase the wildlife harvest and shrink the stock if given the management control. This result adds important insight into wildlife management systems and suggests that project managers should implement local control with care in areas where the wildlife induces damage to agricultural crops.

This paper supports the theoretical modelling with a numerical analysis of the wildebeest exploitation in Serengeti. The numerical considerations indicate that local property rights may work as an incentive to deplete the wildebeest stock. It turns out that a better conservation strategy for Serengeti is to support efforts which protect crops from roaming wildlife.

6.3. Designing integrated conservation and development projects: Hunting incentives and human welfare with numerical illustrations from Serengeti

As mentioned, chapter 3 investigates the impact on wildlife conservation of a common property management regime where the local people constitute the group of co-owners. There are, however, other ways in which the local community can participate in wildlife management. As discussed in section 3, participation may take place through benefit sharing. This is covered in chapter 4, where the local people are considered to have no property rights to wildlife, but benefit from money transfers from tourism and distribution of game meat from managed harvests. This chapter differs from chapter 2 and 3 in the following ways. First, wildlife migration is not of focus here. Instead, the wildlife stock is roaming within a protected area. Second, while the State has the property rights to wildlife, these rights are not effectively protected as illegal hunting occurs in the protected area. The final and major difference compared to chapters 2 and 3, is that this paper considers the local people as a group of households who are both consumers and producers of game meat and agricultural output. The household model links the producer and consumer decision and, by doing so, it captures some important impacts of benefit participation which otherwise would have been lost. In a model of a firm, benefit transfers works as lump-sum income and will not affect the decision to hunt. However, when modelling the agent as a household, such transfers may affect the decision to hunt by altering the budget constraint.

As discussed in section 3, there are some possible pitfalls of management systems relying on benefit transfers only, one of which is the lack of a link between illegal wildlife utilization and the transfers. Chapter 4 highlights this shortcoming by demonstrating that the project's design is detrimental to the performance of a benefit-sharing project. First, the theoretical results suggest that stimulating work opportunities in tourism has the potential of promoting wildlife conservation and human welfare. In contrast, money transfers from tourism and distribution of game meat from managed harvests fail if not explicitly linked to the conservation objective. This is in accordance with the analysis presented by Barrett and Arcese (1998). In the next step, such a link is modelled by introducing a risk of being excluded from the project benefits if caught in illegal hunting. The model demonstrates that relying on this kind of link may be a durable mean to reach the goal of improved wildlife conservation and human welfare.

This essay gives a numerical analysis of the wildlife exploitation in Serengeti. The simulations reveal some interesting features compared to the results of Barrett and Arcese (1998). While they claim that game meat distribution will lead to wildlife degradation in Serengeti, I show that game cropping up to a certain level, and in presence of a proper link, will promote wildlife conservation in this area.

6.4. Wildlife conservation policies and incentives to hunt: An empirical analysis of illegal hunting in western Serengeti, Tanzania

The last paper of this thesis provides an empirical analysis of the incentives to hunt among local people living on the western border of the Serengeti National Park. This paper estimates functions for labour use in illegal hunting by using cross-sectional survey data on the household level. Data were collected through household interviews based on a questionnaire on economic conditions such as illegal hunting, income, land use and crop composition in agriculture, domestic animal keeping, and wildlife-induced damage. In addition, the survey captures participation in the benefit-sharing project established in the study area, namely the Serengeti Regional Conservation Project (SRCP).

The purpose of this paper is to investigate the impact on illegal hunting of participation in SRCP, the pattern of crop production, domestic animal keeping, and wildlife-induced damage to crops and domestic animals. The results show that hunting effort is inversely related to participation in SRCP. In addition, the type of crops produced in agriculture seems to be detrimental to the hunting effort. Households cultivating a high share of cash crops are likely to spend less time hunting. In addition, policies aimed at preventing wildlife-induced damage to domestic animals have the potential of reducing the hunting pressure. When it comes to reported income, it seems that SRCP fails to promote economic development among the local people. Instead, the analysis suggests that policies stimulating cash crop production and encouraging households to put crops on the market represent a better tool for promoting both wildlife conservation and human welfare.

7. Some concluding remarks and steps forward

The basic idea behind this thesis is to utilize bio-economic modelling to better understand the incentives for exploiting wildlife, and to analyse how various wildlife management programmes influence these incentives. The essays show that there is no easy approach for promoting wildlife conservation. Gazetting land may under some circumstances increase the wildlife stock, but may under other circumstances trigger the depletion of wildlife. A rule of thumb seems to be that habitat protection may have unintended effects in areas where cultivated land is withdrawn from people who rely heavily on agricultural production as a source of income. Protected areas are more successful when non-cultivated land is gazetted.

However, with low costs of living with wildlife, the conservation interest of the local people is likely to coincide with the interest of the project management authority. This may lead to success for a co-operative local management regime. In contrast, communities experiencing high costs of living with wildlife may fail in conserving the resource. Instead, benefit-sharing schemes with a proper link to the conservation objective may encourage wildlife conservation and improve the economic conditions for the local people. Finally, the empirical investigation indicates that stimulating an income-generating composition of crops in agriculture is a promising way of enhancing both conservation and welfare.

The four essays offer valuable insight into the analysis of wildlife utilization and management, but undeniably, there is still a lot of research to be done in this area. First, all models in this thesis consider a fixed human population. However, it is easy to picture that benefit-sharing projects and local property rights may attract people from outside the local community and lead to immigration to these areas. Therefore, bio-economic models with endogenous population growth should be explored. How will immigration affect the co-operation between people in the local community, and between the local people and the project authority? How does it affect the land use and hence, the exploitation of wildlife and its resource base? Second, wildlife is considered one single species throughout this thesis. However, the real world consists of interacting species, such as predator and preys, species competing for grazing areas and so forth. Obvious issues regarding this shortcoming are of importance when investigating wildlife utilization and management.

This thesis explains possible bio-economic outcomes using partially equilibrium models. That is, the local people are considered the only active agent, while the park authority is passive. The analyses are positive and, hence, they do not predict how the local people and the park manager should operate in order to reach a social optimum. Both models involving an active park manager and normative analyses of wildlife management are necessary in order to reveal the challenges and to design future management strategies. Such models should also include game theory to capture the interaction between the park manager and the local people. For instance, when analysing the design of benefit-sharing schemes, this thesis assumes that the park manager defines ‘poachers’ as those caught in illegal hunting, whereas people who manage to escape are defined as non-poachers. However, in order to design the optimal benefit-sharing scheme, the challenge is to set up prospects consisting of benefit transfers and anti-poaching law enforcement in such a way that the poachers who would otherwise manage to escape will choose the benefit transfers instead of illegal hunting.

In order to design future management strategies, further research should also involve empirical studies. Here, emphasis should be placed on investigating the impact of economic and social conditions on the incentives for exploiting natural resources. In addition, case studies of existing management systems offer valuable information on project designs. There is no representative ‘model’ of wildlife management and therefore, each and every project must address the most serious conservation and development issues in the particular geographical area. The challenge for scientists is to provide the knowledge required to design the ultimate management strategy for the region in question.

References

Akerlof, G. A. (1984): *An Economic Theorist's Book of Tales*, Cambridge University Press, Cambridge.

Baland, J.M. and J.P. Platteau (1996): *Halting Degradation of Natural Resources*, Clarendon Press, Oxford.

Barbier, E.B. (1992): "Community-based development in Africa". In Swanson, T.M. and E.B. Barbier (eds.): *Economics for the Wilds: Wildlife, Wildlands, Diversity and Development*, Earthscan, London, 103-135.

Bardhan, P. (1993): "Symposium on management of local commons", *Journal of Economic Perspectives*, 7(4), 87-92.

Barrett, C.B. and Arcese, P. (1995): "Are integrated conservation-development projects (ICDPs) sustainable? On the conservation of large mammals in sub-Saharan Africa", *World Development*, 23(7), 1073-1084.

Barrett, C.B. and P. Arcese (1998): "Wildlife harvest in integrated conservation and development projects: Linking harvest to household demand, agricultural production, and environmental shocks in the Serengeti", *Land economics*, 74(4), 449-465.

Brandon, K.E. and M. Wells (1992): "Planning for people and parks: design dilemmas", *World Development*, 20(4), 557-570.

Bromley, D. (1991): *Environment and Economy*, Blackwell, Oxford.

Brown, G. (2000): "Management of wildlife and habitat in developing countries". In Dasgupta, P. and K-G Mäler (eds.): *The Environment and Emerging Development Issues*, Oxford University Press, Oxford, 554-573.

Bulte, E.H. and G.C. van Kooten (1996): "A note on ivory trade ban and elephant conservation", *Environment and Development Economics*, 1(4), 433-443.

Bulte, E.H. and G.C. van Kooten (1999): "Economics of antipoaching enforcement and the ivory trade ban", *American Journal of Agricultural Economics*, 81(2), 453-466.

Campbell, K. and H. Hofer (1995): "People and wildlife: Spatial dynamics and zones of interaction". In Sinclair, A.R.E. and P. Arcese (eds): *Serengeti II*, University of Chicago Press, Chicago, 534-570.

Carlson, G.A. and M.E. Wetzstein (1993): "Pesticides and pest management". In Carlson, G.A., D. Zilberman, and J.A. Miranowski (eds): *Agricultural and Environmental Resource Economics*, Oxford University Press, Oxford, 268-318.

Clark, C.W. (1976): *Mathematical Bioeconomics*, Wiley, New York.

Clark, C.W. (1990): *Mathematical Bioeconomics*, second edition, Wiley, New York.

Clark, C.W. (1973): "Profit maximization and the extinction of animal species", *Journal of Political Economy*, 81(4), 950-961.

Clark, C.W. and G.R. Munro (1975): "The economics of fishing and modern capital theory: A simplified approach", *Journal of Environmental Economics and Management*, 2, 92-106.

Conrad, J.M. (1998): "The bioeconomics of marine sanctuaries", *Journal of Bioeconomics*, 1(2), 205-217.

Dasgupta, P. (1993): *An Inquiry into Well-Being and Destitution*, Clarendon Press, Oxford.

Dasgupta, P. and K-G Mäler (1995): "Poverty, institutions, and the environmental resource-base". In Behrman, J. and T.N. Srinivasan (eds): *Handbook of Development Economics*, 3, Elsevier, Amstredam.

Doran, M.H., A.R.C. Low and R.L. Kemp (1979): "Cattle as a store of wealth in Swaziland: Implications for livestock development and overgrazing in eastern and southern Africa", *American Journal of Agricultural Economics*, 61(1), 41-47.

Elster, J. (1989): *Nuts and Bolts for the Social Sciences*, Cambridge University Press, Cambridge.

Emerton, L. (1998): "Why wildlife conservation has not economically benefited the communities in Africa", *Community Conservation Research in Africa*, Paper no 5, Institute for Development Policy and Management, University of Manchester, Manchester.

Gibson, C.C. and S.A. Marks (1995): "Transforming rural hunters into conservationists: An assessment of community-based wildlife management programs in Africa", *World Development*, 23(6), 941-957.

Gibson, C.C. (1999): *Politicians and Poachers: The Political Economy of Wildlife Policy in Africa*. Cambridge University Press, Cambridge.

Gordon, H.S. (1954): "The economic theory of a common property resource: the fishery", *Journal of Political Economy*, 80, 124-142.

Gravelle, H. and R. Rees (1992): *Microeconomics*, Longman Group UK Ltd, London.

Hannesson, R. (1998): "Marine reserves: what do they accomplish? ", *Marine Resource Economics*, 13(3), 159-170.

Hardin, G. (1968): "The tragedy of the commons", *Science*, 162, 1243-1248.

Hofer, H., K. Campbell, and M.L. East (2000): "Modeling the spatial distribution of the economic costs and benefits of illegal game meat hunting in the Serengeti", *Natural Resource Modeling*, 13(1), 151-177.

Homans, F.R. and J.E. Wilen (1997): A model of regulated open access resource use. *Journal of Environmental Economics and Management*, 32(1), 1-21.

Huffaker, R.G., M.G. Bhat, and S.M. Lenhart (1992): "Optimal trapping strategies for diffusing nuisance-beaver populations", *Natural Resource Modeling*, 6(1), 71-97.

Kiss, A. (1990): "Living with Wildlife: Wildlife Resource Management with local Participation in Africa", *Technical Paper*, 130, World Bank, Washington D.C.

Leader-Williams, N., S.D. Albon and P.S.M. Berry (1990): "Illegal exploitation of black rhinoceros and elephant populations: patterns of decline, law enforcement and patrol effort in Luangwa Valley, Zambia", *Journal of Applied Ecology*, 27(3), 1055-1087.

Marks, S.A. (1984): *The Imperial Lion: Human Dimensions of Wildlife Management in Africa*. Westview Press, Boulder.

Mesterton-Gibbons, M. (1993): "Game-Theoretic Resource Modeling", *Natural Resource Modeling*, 7(2), 93-147.

Milner-Gulland, E.J. and N. Leader-Williams (1992a): "A model of incentives for the illegal exploitation of black rhinos and elephants: poaching pays in Luangwa Valley, Zambia", *Journal of Applied Ecology*, 29(2), 388-401.

Milner-Gulland, E.J. and N. Leader-Williams (1992b): "Illegal exploitation of wildlife". In Swanson, T.M. and E.B. Barbier (eds): *Economics for the Wilds: Wildlife, Wildlands, Diversity and Development*, Earthscan, London, 195-213.

Munro, G.R. and A.D. Scott (1985): "The economics of fishery management", *Handbook of Natural Resource and Energy Economics*, 2, 623-676.

Nepal, S.K. and K.E. Weber (1995): "Managing resources and resolving conflicts: national parks and local people", *International Journal of Sustainable Development and World Ecology*, 2, 11-25.

Ostrom, E. (1990): *Governing the Commons*, Cambridge University Press, Cambridge.

Perrings, C. (1993): "Stress, shock and the sustainability of optimal resource utilization in a stochastic environment". In Barbier, E.B. (ed): *Economics and Ecology: New Frontiers and Sustainable Development*, Chapman & Hall, London, 66-95.

Pezzey, J., C.M. Roberts and B.T. Urdal (2000): "A simple bioeconomic model of a marine fishery reserve", *Ecological Economics*, 33(1), 77-91.

Plourde, C.G. (1970): "A simple model of replenishable natural resource exploitation", *American Economic Review*, 60(3), 518-522.

Reid, W.V. and K.R. Miller (1989): *Keeping Options Alive: The Scientific Basis for Conserving Biodiversity*, World Resources Institute, Washington DC.

Sanchirico, J.N. and J.E. Wilen (2001): "A bioeconomic model of marine reserve creation", *Journal of Environmental Economics and Management*, 42(3), 257-276.

Schulz, C.E. and A. Skonhoft (1996): "Wildlife management, land-use and conflicts", *Environment and Development Economics*, 1(3), 265-280.

Seabright, P. (1993): "Managing the local commons: theoretical issues in incentive design", *Journal of Economic Perspectives*, 7(4), 113-134.

Sinclair, A.R.E. and P. Arcese (1995): *Serengeti II*, University of Chicago Press, Chicago.

Skonhoft, A. (1999): "On the optimal exploitation of terrestrial animal species", *Environmental and Resource Economics*, 13(1), 45-59.

Skonhoft, A. (1998): "Resource utilization, property rights and welfare. Wildlife and the local people", *Ecological Economics*, 26(1), 67-80.

Skonhoft, A. and J.T. Solstad (1998a): "Investing in wildlife: can wildlife pay its way?", *Journal of African Economies*, 7(2), 237-262.

Skonhoft, A. and J.T. Solstad (1998b): "The political economy of wildlife exploitation", *Land Economics*, 74(1), 16-31.

Smith, A.B. (1992): *Pastoralism in Africa: Origins and Development Ecology*, Hurst & Company, London.

Smith, V.L. (1969): "On models of commercial fishing", *Journal of Political Economy*, 77(2), 181-198.

Smith, V.L. (1975): "The primitive hunter culture, pleistocene extinction, and the rise of agriculture" *Journal of Political Economy*, 83(4), 727-755.

Songorwa, A.N. (1999): "Community-based wildlife management (CWM) in Tanzania: Are the communities interested?", *World Development*, 27(12), 2061-79.

Swanson, T.M. (1994): "The economics of extinction revisited and revised: a generalized framework for the analysis of endangered species and biodiversity losses", *Oxford Economic Papers*, 46, 800-821.

Swanson, T.M. and E.B. Barbier (1992): *Economics for the Wilds: Wildlife, Wildlands, Diversity and Development*, Earthscan, London.

Wells, M. (1992): "Biodiversity conservation, affluence and poverty: Mismatched costs and benefits and efforts to remedy them", *Ambio*, 21(3), 237-243.

Wells, M. and K. Brandon (1992): *People and Parks: Linking Protected Area Management with Local Communities*, World Bank, World Wildlife Fund and U.S. Agency for International Development, Washington DC.

Zivin, J., B.M. Hueth and D. Zilberman (2000): "Managing a multiple-use resource: The case of feral pig management in California rangeland", *Journal of Environmental Economics and Management*, 39(2), 189-204.

Chapter 2

Wildlife conservation, human welfare and the failure of protected areas

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Abstract

The establishment and expansion of protected areas in Africa have been motivated by the aspiration of increased wildlife abundance. However, the increasing poaching pressure on this continent has led to the perception that protected areas have failed in preserving wildlife. This paper presents a bio-economic model in order to explain the economic factors and mechanisms which may have caused this failure. The analysis focuses on a hunter-agrarian economy where an opportunity cost of habitat protection, due to less land for agricultural cultivation, pasture, and wildlife hunting, is present. An expansion of the protected area restricts the local people's user rights to land. Depending on the economic conditions in agriculture and wildlife hunting, this policy may reduce the degree of wildlife conservation. This result contrasts the theoretical findings of fishery economics, where a marine reserve increases the aggregate fish stock. In addition to the conservation effect of protected areas, this paper investigates the impact on human welfare of restricting the user rights to wildlife and land.

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1. Introduction

The initial approach to preserve natural resources in Africa had its roots in the Western environmentalist movement of the 20th century. This approach saw the establishment of large areas of national parks and reserves as the foremost priority for African conservation (Marks 1984, Kiss 1990, Swanson and Barbier 1992). The objective of this management system was to protect wild animals and natural habitats through prohibition or restriction of wildlife utilization. Setting aside areas for national parks and game reserves is still the predominating management strategy (Swanson and Barbier 1992). The control and management of protected areas are usually vested in the State, which reaps economic benefits from wildlife tourism. In contrast, gazetting land for wildlife protection has displaced rural communities and curtailed their access to natural resources that they previously had access to. Land for cultivation and pasture has been lost and harvesting of wildlife in these areas has been deemed illegal (Marks 1984, Kiss 1990, Swanson and Barbier 1992, Wells 1992). In addition, local communities bear the costs of living with wildlife through agricultural damage induced by animals roaming on agricultural land. Hence, while the State reaps the benefits of protected areas, the costs are borne at the local level.

The idea of protected areas was motivated by the aspiration of increased wildlife abundance. The continuing expansion of protected areas in Africa reflects that this perception is still prevalent. However, the increasing poaching pressure has led to a growing recognition that protected areas have failed in their goals of preserving wildlife (Kiss 1990, Swanson and Barbier 1992, Martin 1993, Barrett and Arcese 1995, Gibson and Marks 1995, Songorwa 1999). Martin (1993), for instance, discusses protected areas as a tool in wildlife conservation. He claims that Africa has made the mistake of gazetting too many and too large areas to be able to meet the minimum operating costs required in order to conserve and protect wildlife in these areas. He pictures an inevitable situation where budgets are to small to prevent illegal exploitation, leading all areas to deteriorate simultaneously (see also Leader-Williams and Albon 1988, Dixon and Sherman 1991). Instead of focusing on law enforcement and the amount of protected land, Kiss (1990) and Swanson and Barbier (1992), among others, point to the lack of economic compensation to the local people as an explanation of the failure of protected areas. They argue that it is necessary to correct this distortion in order to promote wildlife conservation, and suggest that this is achievable through revenue sharing in wildlife related activities. By providing the local people with such benefits, they believe that the management authorities will gain the co-operation of the local people and thus reduce their incentives to exploit wildlife.

None of the authors cited above have adopted a model-theoretical framework to explore the conservation-effect of protected terrestrial habitats. In contrast, in fishery economics, marine reserves have been analysed in a bio-economic context by Conrad (1998), Hannesson (1998) Pezzey et al. (2000), and Sanchirico and Wilen (2001), among others. Sanchirico and Wilen (2001) consider two fishing patches, initially characterised as open-access fisheries (entry until zero rents). A marine reserve is created by closing one patch for fishing. The fish stock in the open patch is determined by a fixed cost-price ratio and is not altered by closing the other patch. Based on these assumptions, a marine reserve increases the aggregate biomass of the two patches for every ecological system. Sanchirico and Wilen (2001) also focus on the economic impact of a marine reserve. Because free access to the open patch means zero rent, they define the fishery as better off if a marine reserve increases the total harvest. As the fish stock disperses between the patches, they show that the effect on total harvest of closing one patch is positive if increased dispersal between the reserve and the open patch compensates for the foregone harvest in the reserve. Also Hannesson (1998) shows that marine reserve creation increases the aggregate fish stock when there is open access to the area outside the reserve. However, he demonstrates that a marine reserve of a moderate size will have only a small conservation effect, compared with open access to the entire area inhabited by the stock. In addition, Hannesson (1998) shows that the impact on the aggregate catch depends on the size of the marine reserve.

This paper adds to the research of marine reserves by presenting a bio-economic model of wildlife habitat protection. Terrestrial habitats differ from marine habitats in that there may be, in addition to hunting, an alternative use of protected land. In the present, agricultural crop production is considered the alternative use of land. In order to draw a line to marine reserves, this paper makes a distinction between two policies of land protection. The difference between these policies lies in the *type* of land gazetted. One alternative is to establish a protected area by gazetting non-cultivated land only. In such a case, there is no alternative use of the protected area except hunting. This policy is therefore quite similar to marine reserve creation and the analysis demonstrates that it promotes wildlife conservation. However, rapid human population growth in Africa has forced humans to bring their agricultural activities ever more close to wildlife habitats (see Dixon and Sherman 1991 and Martin 1993). The second alternative is therefore to expropriate cultivated land for wildlife protection. In this case the protected area does not only close off an area for hunting, it also withdraws land previously used in agriculture. Consequently, an alternative cost of habitat protection is present, namely the foregone return from crop production. The analysis shows, in contrast to marine reserves, that this policy may cause wildlife degradation. Analysing the driving forces behind such an outcome and its impact on human welfare are the main contributions of this paper.

The bio-economic model developed in this paper draws on the biological system presented by Hannesson (1998). See also Armstrong and Reithe (2001). Hannesson looks at a fish stock located in an area of a fixed size and the marine reserve is defined as a subset of this area. In the present paper the biomass is a wildlife stock dispersing over a fixed area or ecosystem. The ecosystem contains two sub-areas, the protected area and the outer area. The protected area is managed by an agency appointed by the State. On the border to the park a group of peasants utilize the outer land for agricultural production and wildlife hunting. Following the fishery analyses cited above, it is assumed that hunting is not allowed within the protected area and that law enforcement is effectively preventing illegal hunting here. However, the local people have legal rights to exploit the land in the outer area and the wildlife roaming outside the park. That is, they have user rights to land and wildlife in the outer area[1].

Throughout this analysis the local people are considered the only active agent. It is assumed that the size of the protected land is determined by the State and, therefore,

[1] In a state property regime individuals or groups may be allowed to make use of the natural resources without having any property rights. Bromley (1991) defines this as usufruct rights.

cannot be altered by the park manager. Hence, expropriation of land will be considered exogenous in the model. In addition, because we neglect illegal hunting, there is no need to focus on law enforcement and consequently, the park manager plays no role in the subsequent model. Instead, the manager is a passive beneficiary of non-consumptive use of the protected stock, for instance tourism[2].

The rest of this paper is organized as follows. Section 2 presents the ecological model, while the behaviour of the local people follows in section 3. The impact on wildlife conservation and human welfare of protected area creation is investigated in section 4. A summary and discussion follows in section 5.

2. The ecological model

Consider an area or ecosystem of fixed size divided in two sub-areas; a protected area and an outer area. The ecological modelling is identical to Hannesson (1998) who looks at species dispersing between the sub-areas in a density-dependent way. This means that wildlife migrate to the relatively less dense area (see e.g. Pulliam 1988). The animals roam freely between the sub-areas, because there are no physical obstructions, e.g. fencing, separating the parkland from the open area. It is further assumed, as already mentioned, that wildlife harvesting only takes place when the species are outside the protected area.

In the following, some restrictive assumptions are made about the quality of land. First, land is considered homogenous, i.e. every part of the ecosystem is equally suitable as habitat for wildlife. Secondly, although agricultural production takes place in the outer area, we assume no incompatibility in land use. That is, there is no negative impact on the living conditions of wildlife of adding more land to agricultural production. However, in reality, unexploited land may generate more wildlife than agricultural land as land clearing, fencing and so forth result in poorer conditions and smaller refuges for wildlife (see Norton-Griffiths 2000). This may be captured, as in Huffaker et al. (1992), by assuming a smaller intrinsic growth rate of wildlife in the outer area. Skonhoft

[2] It is therefore assumed that the management of the protected area involves no harvesting or culling. For a critical review of this assumption, see e.g. Wright (1999).

(1999) and Bulte and Horan (2000) present an alternative approach where the carrying capacity in the outer area is specified as a decreasing function of the amount of land utilized in agricultural production or other alternative uses. However, in order to capture the main ideas, no incompatibles in land uses are assumed to be present here.

The purpose of this paper is to analyse the conservation effect of altering the size of the protected sub-area. An increase in the size of this area is followed by the same reduction in the outer area. Therefore, the ecological part of the model, which is identical to the system presented by Hannesson (1998), specifies the migration rates between the sub-areas as dependent on the size of the protected area. Technically, the probability of an animal being located in the protected area or the outer area equals the size of the respective areas. Now, assume that the size of the ecosystem is normalized to one. A fraction w of this area is gazetted as protected land and consequently, $(1-w)$ is the size of the outer area. Let $X(t)$ be the density of the stock in the protected area at time t, while $Y(t)$ is the density in the outer area at time t. In the following, the time subscript is omitted. The size of the wildlife stock in the protected area and the outer area is wX and $(1-w)Y$ respectively, so that the aggregate stock equals $S = wX + (1-w)Y$.

Let $z \geq 0$ be the moving rate of wildlife, i.e. the rate at which an animal moves to bring it to the nearest suitable spot for grazing or prey[3]. $z = 0$ means that the animals do not move around at all. The rate of dispersal of the stock in the protected area is then zwX. $(1-w)$ is the probability that the moving animal will migrate out of the reserve. The migration out of the reserve is therefore $z(1-w)wX$. To translate this into change in stock *density* in the outer area, we divide it by the size of that area. Hence, the increase in the density of wildlife in the outer area due to migration from the protected area is zwX. Similarly, $zw(1-w)Y$ is the migration from the outer area onto protected land. The reduction in the density of wildlife in the outer area due to migration to the conservation area is then zwY. In the same way, the change in the stock density in the conservation

[3] The moving rate may also be related to breeding. For instance, animals with slow growing non-precocial young are obliged to stay within a small area to breed. This is the case for carnivores like lions and hyenas. In contrast, ungulates with precocial young do not need to stay in one place because the young can follow the mother within an hour or so of birth (Caughley and Sinclair 1994).

area due to migration from the outer area is $z(1-w)Y$, while the stock density in the conservation area is reduced by $z(1-w)X$ due to migration to the outer area.

Because of the non-incompatibility of land, the carrying capacity per square kilometre is equal in each sub-area and therefore normalized to one. Natural growth is assumed to take place in both sub-areas and is given by a logistic growth function. The rate of change in the density of wildlife in the two sub-areas is given by[4]

$$(1) \qquad dX/dt = rX(1-X) + z(1-w)(Y-X)$$

$$(2) \qquad dY/dt = rY(1-Y) + zw(X-Y) - h$$

Here, h is the harvesting rate, while r is the intrinsic growth rate. Note that the rate of change in the aggregate stock is given by $dS/dt = wdX/dt + (1-w)dY/dt$. If the whole ecosystem is gazetted for wildlife protection ($w=1$), then $S = X$ and $dS/dt = rS(1-S)$. In the same way, with no protection ($w=0$) $S=Y$ and $dS/dt = rS(1-S) - h$. Throughout the analysis it is assumed that $0 < w < 1$.

In absence of man, $h = 0$, Figure 1 illustrates the isoclines of *(1)* and *(2)*. This figure is quite similar to the graphical demonstration of a two-patch density-dependent system in Skonhoft (1999) and Sanchirico and Wilen (2001) (see also Appendix 1). Here, the marginal migration rates are below the maximum specific growth rate so that $1 - zw/r > 0$ and $1 - z(1-w)/r > 0$. This makes sense because a system with a migration exceeding the intrinsic growth is likely to fail in sustaining an ecological equilibrium with positive biomass within each patch. The *X*-isocline is a strictly convex function of *X* and runs through the point (1,1). Above the isocline, the natural growth and dispersion from the outer area exceed the dispersion out of the reserve so that $dX/dt > 0$. The opposite occurs below the isocline. The *Y*-isocline is a strictly concave

[4] The dispersal functions of *(1)* and *(2)* are somehow different from those presented by Conrad (1998), Skonhoft (1999) and Sanchirico and Wilen (2001). This is explained and justified in Appendix 1.

function of X and runs through the point (1,1). Below the isocline, dY/dt is positive, whereas above, dY/dt is negative.

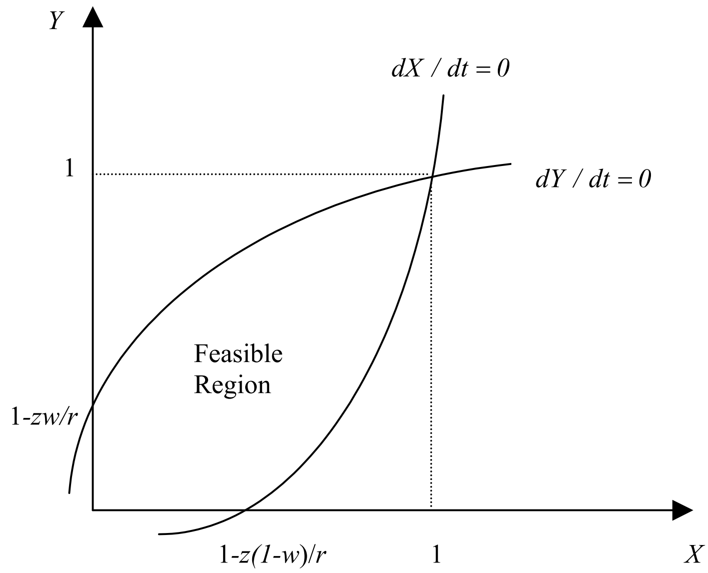

Figure 1: Ecological equilibrium in absence of man.

In absence of man and migration below the intrinsic growth, it will therefore be a unique equilibrium with stock densities equal to one. Hence, the aggregate stock equals one in equilibrium. It can be demonstrated that the equilibrium is stable[5]. The feasible region for an interior solution of the system is found in the area closed by the isoclines and the axes. The size of this region depends on the biological parameters of the model. If the moving rate z approaches zero, i.e. a system of closed and independent patches, the individual stocks collapse to zero or the carrying capacity of its area. If the moving rate increases so that zw/r (or $z(1-w)/r$) approaches one, the feasible region reduces and collapses to a lens with intersection at (a,0) (or (0,b)) and (1,1), where $a = 1 - z(1-w)/r$ (and $b = 1 - zw/r$).

Throughout this analysis it is assumed that the patches are interdependent, i.e. z is positive. Introducing human activity as a fixed positive harvesting rate in this system

[5] The stability conditions read $\partial f(1,1)/\partial X + \partial g(1,1)/\partial Y = -(2r+z) < 0$ and $(\partial f(1,1)/\partial X)(\partial g(1,1)/\partial Y) - (\partial f(1,1)/\partial Y)(\partial g(1,1)/\partial X) = r(r+z) > 0$.

shifts the Y-isocline in Figure 1 down, i.e. human activity reduces the density in the outer area for a given stock density in the game reserve. Consequently, due to a relative dense population in the protected area, wildlife disperses to the outer area, which causes a decline in X. This illustrates that harvesting in the outer area spells over to the protected area. The system settles in a new stable equilibrium where both stock levels are smaller than their respective carrying capacities and $Y < X$. Throughout the remaining analysis it is assumed that the system is in ecological equilibrium ($dX / dt = dY / dt = 0$).

3. The economy

The ecological steady state above was established for a given harvesting rate. However, the harvesting activity is determined by economic considerations, which are outlined in this section. Before we move to the economic part, it is convenient to establish the different ways in which land is utilized in this model. Recall that land is utilized by two agents. First we have the park manager who utilizes the protected area in production of non-consumptive tourism services (see section 1). Second, we have the local people who have legal rights to utilize the outer area in agricultural production and wildlife hunting.

The State may instruct the park manager, for presumed conservation purposes, to expand the protected area. This can only take place by implementing parts of the outer land into the park area. There are two ways in which the State may accomplish this, and these are related to the type of land as discussed in section 1. First, if present, the State can protect non-cultivated land. For the local people living in the outer area, this policy represents limited user rights to wildlife, but no restriction on the rights to exploit land already cultivated for agricultural use. Technically, this will be the case where the constraint on agricultural land is non-binding. Second, in marginal areas, the State must expropriate cultivated land in order to expand the protected area. For the local people, this procedure restricts their user rights to agricultural land as well as their user rights to wildlife. This will be the case when the constraint on agricultural land use is binding. The two scenarios of protected area expansion will be analysed in section 4.1 and 4.2, respectively.

The next step is to present a formal model of the hunting and agricultural decision of the local people. Throughout the analysis the local people are considered a homogenous group of peasants and, in line with traditional reasoning, it is assumed that the elders are in charge of the group's activities (Marks 1984). The number of animals harvested H is specified as an increasing function of labour effort E_h, stock density Y, and the size of the outer area $(1-w)$, as $H = H(E_h, Y, 1-w)$. In the following, H is considered linear in E_h and Y, but concave in $(1-w)$. The wildlife offtake is specified as

$$(3) \qquad H = f(1-w)E_h Y$$

Here, $f(1-w)$ is interpreted as the catchability coefficient. $f' > 0$ because additional areas are open for hunting as the outer area expands[6]. $f'' < 0$ reflects that the marginal catchability decreases with the size of the hunting ground due increased distance from the home region and because a larger area is likely to include a more diverse and less advantageous topography. To translate the offtake into change in the wildlife density in the outer area in *(2)*, we divide H with the size of this area, so that $h = H/(1-w)$ is the hunting rate[7].

The next step is to present the agricultural activity, interpreted as crop production, of the local people. The agricultural production is a function of labour E_A and land L as

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[6] This formulation of H differs from Skonhoft (1999) and Pezzey et al. (2000) who use a Schaefer specification $H = qE_h y/K_y$, where q is the catchability coefficient, y is the stock size, and K_y is the carrying capacity of the outer area. They further assume that K_y is an increasing function of the size of the outer area. That is, in contrast to the present model, and for a given E_h, the wildlife offtake decreases with the size of this area due to reduced stock density. However, the ecological system presented by Skonhoft and Pezzey et al. differs from the present system in that they allow for increases in the size of the outer area without altering the size of the protected land (see Appendix 1). Then, by assuming that wildlife roams onto the 'new' land, an increase in the habitat size in the outer area has a direct negative effect on stock density y/K_y, for a given stock size y. Hence, they assume that the impact on wildlife offtake is negative, simply because a less dense population is harder to catch. Here, however, there is no *direct* effect on stock density Y of increasing the size of the outer area. That is because the outer area increases by altering a non-physical border between the protected land and the outer land.

[7] Following Barrett and Arcese (1998), Lopez (1998), Skonhoft (1998), Skonhoft and Solstad (1998), and Bulte and van Soest (1999), the local people are the only agents involved in wildlife hunting in the outer area. That is, we ignore the possibility of outsiders and professional gangs area entering this area for hunting.

$A(E_A, L)$ (see below). The endowment of labour is normalized to one and, hence, the constraint on labour use reads

$$(4) \qquad E_h + E_A \leq 1,$$

Throughout the analysis it is assumed that the constraint is binding. A trade-off between wildlife hunting and agricultural production is present in that the opportunity cost of wildlife harvesting equals the foregone return from agricultural production (and vice versa).

As mentioned in section 2, land is homogeneous as habitat for wildlife. It is therefore convenient to consider land as homogeneous for agricultural uses as well. This means that additional land is equally suitable in agriculture as previously exploited land (see also Bulte and Horan 2000). Then, proportional increases in labour effort and land use must cause output to increase by the same proportion. Consequently, the average returns to land A/L and labour A/E_A are left unchanged. The agricultural production function is therefore characterised by constant returns to scale and specified as a Cobb-Douglas type as follows (Hayami and Ruttan 1985).

$$(5) \qquad A(E_A, L) = \mu E_A^{\alpha} L^{1-\alpha},$$

Here, $\mu > 0$ is a technology parameter and $0 < \alpha < 1$ is the output elasticity of labour. Because of its homogeneity, diminishing return to land is not caused by taking inferior land into production, but by reduced labour effort per unit of land[8]. The total area available for agricultural production is given by the size of the outer area $(1-w)$. The constraint on land use is therefore given by

$$(6) \qquad L \leq (1-w)$$

[8] For linear homogeneous or constant return to scale production functions, the marginal products are independent of scale and depend only on the input proportions.

Investment costs on land, for instance related to clearing and fencing, and costs in cultivation and crop harvesting are ignored in this analysis[9]. The only agricultural cost of consideration here is related to damage caused by wildlife roaming on agricultural land. The nuisance stream per unit of land is equal to cY, with $c > 0$ and fixed[10]. Consequently, the total damage of the wildlife roaming on agricultural land is cLY. c is interpreted as the marginal damage per animal. All else equal, more agricultural land means more nuisances.

When inserting for the effort constraint *(4)* into the production function in *(5)*, the net benefit function of the local people yields

$$(7) \qquad \pi = P_h f(1-w)E_h Y + P_A \mu(1-E_h)^{\alpha} L^{1-\alpha} - P_A cLY ,$$

where P_h and P_A denote the price of game meat and agricultural output, respectively[11]. These prices are assumed fixed throughout the analysis.

As mentioned, the local people have user rights to land and wildlife. This means that they are not granted titles to these resources and, consequently, they face a continuing risk of the State withdrawing their user rights through an expansion of the protected area. The local people have therefore few, if any, incentives to base their wildlife harvesting on long-term considerations. Hence, they do not take the stock of wildlife into account when deciding upon their effort use[12]. Technically, this is captured by assuming that the local peasants treat the stock density Y as exogenous, which is in accordance with one of Smith's models (1975). See also Skonhoft and Solstad (1998). The local people choose the hunting effort E_h and cultivated land L to maximize *(7)*,

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[9] In reality, however, there will be costs related land clearing, fencing, sowing, fertiliser and pesticide use and so forth. Modelling such costs will not alter the qualitative results of this analysis.

[10] In reality, the local people can perform damage control through fencing, guard patrols and so forth. In the model this would have worked through a changing *c*. Here, such measures are neglected.

[11] In accordance with the traditions in the past century, it is therefore assumed that no economic compensation is paid to the local people for the loss of access to land and wildlife (Marks 1984, Kiss 1990, Swanson and Barbier 1992, Wells 1992).

[12] Martin (1993) points out how the risk of land expropriation affects landholders. He writes (p. 15): "The influence of the preservationist lobby is a serious disincentive for the landholder contemplating an investment in wildlife as a land use".

given the constraint on land use in *(6)*. The Lagrange function reads $V = P_h f(1-w)E_h Y + P_A \mu(1-E_h)^{\alpha} L^{1-\alpha} - P_A cLY - \lambda(L-(1-w))$, where λ is the shadow price of land. Equations *(8)-(10)* yield the first order conditions for maximum when an interior solution for hunting effort is supposed to be present.

(8) $\quad P_h f(1-w)Y = P_A \mu\alpha(1-E_h)^{\alpha-1} L^{1-\alpha}$

(9) $\quad P_A \mu(1-\alpha)(1-E_h)^{\alpha} L^{-\alpha} = P_A cY + \lambda$

(10) $\quad \lambda \geq 0;\ \lambda = 0 \text{ if } L < (1-w)$

Equation *(8)* shows that the optimal hunting effort is determined by equality between the marginal product of hunting and the marginal product of labour effort in agricultural production. The decision rule in equation *(9)* states that the local people will convert land in the outer area to agricultural use until the value of the marginal product of land in crop production equals the marginal cost. The marginal cost consists of the value of the marginal damage per unit cultivated land and the shadow value of land. This value equals zero when the constraint on land use is non-binding, while it is positive for a binding constraint (see *(10)*).

The economic equilibrium condition in *(9)* is illustrated graphically in Figure 2. Here, the marginal benefit and costs of land cultivation are measured along the vertical axis. Consider the case of intersection between the marginal cost curve and the marginal benefit curve, which results in $L = L^{1} < (1-w)$. This means that the local people choose not to utilize the whole outer area for cultivation and, hence, $\lambda = 0$. However, a positive shift in agricultural productivity μ and/or a downward shift in the marginal crop damage caused by a lower c or Y, increase the demand for cultivated land. In Figure 2, this is illustrated by an upward shift in the marginal benefit curve caused by a higher μ. For a given land use at L^{1}, the marginal benefit of cultivated land exceeds the marginal crop damage by the positive shadow value of land. The local people respond by converting additional land to agricultural production. In the new equilibrium, λ remains positive if the local people utilize the whole outer area for agricultural production, $L = (1 - w)$, reflecting that land is a scarce factor. This will be the case if μ is 'high', while c and Y are 'low'. In addition, an increase in the size of the protected area w shifts the vertical curve denoting the size of the outer area to the left and increases the shadow value of land.

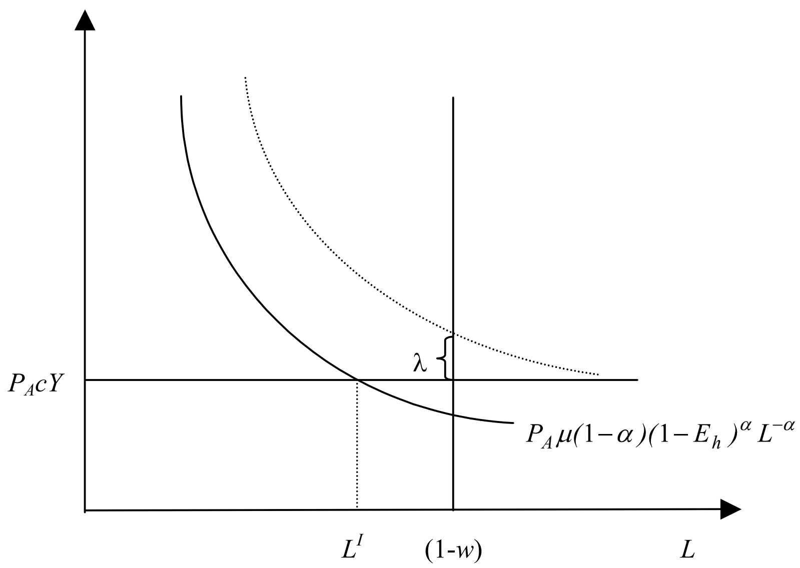


Figure 2: The maximum condition for the amount of cultivated land L. Y and E_h are fixed.

Equation *(8)*-*(10)* together with *(1)* and *(2)* (with $dX / dt = dY / dt = 0$) determine the optimal hunting effort, optimal use of agricultural land and the aggregate stock in ecological equilibrium. The following section describes the two scenarios of a non-binding and a binding constraint on land use.

4. The impact of protected areas on wildlife conservation and local welfare

Above we established the first order conditions maximizing the local people's benefit from wildlife harvesting and agricultural production. In addition, we studied the conditions under which the system settles in a solution where the constraint on land use is binding. The next step is to investigate the impact on wildlife conservation and the welfare of the local people of protected areas. It turns out that the effects are strictly dependent on whether the state gazettes non-cultivated land or expropriates cultivated land, i.e. whether the constraint on land use is non-binding or binding. Section 4.1 considers the case of a non-binding constraint on land use, while the constraint is binding in section 4.2.

4.1 The constraint on land use is non-binding

Assume that the protected area is relatively small, so that land is not a scarce factor in the outer area. Then, the local people settle with an interior solution for cultivated land, $L < (1-w)$, where the marginal return from land equals the marginal damage in *(9)* and $\lambda = 0$. Combining *(8)* and *(9)* (with $\lambda = 0$) and solving for *Y* gives

$$(11) \qquad Y = \mu\left[P_A\alpha / P_h f(1-w)\right]^{\alpha}\left[(1-\alpha)/c\right]^{1-\alpha}$$

Equation *(11)* alone determines the equilibrium stock density in the outer area *Y*. This means that *Y* is determined by the (fixed) economic parameters and the park size only. The result stems on the constant return to scale in the agricultural production function. Accordingly, the input proportion $(1-E_h)/L$ is constant and independent of the scale of crop production for fixed model parameters. This means that we cannot solve the optimal input combination and the resulting crop output from the economic first order conditions. Instead the stock density in the outer area is determined by the economic conditions, while the hunting effort E_h and the stock density in the protected area *X* are solved from the ecological equilibrium in *(1)* and *(2)* (with $dX/dt = dY/dt = 0$). Although the systems are quite different, the same result occurs in Sanchirico and Wilen's model (2001). The aggregate stock follows from $dS/dt = 0$. Finally, the amount of cultivated land *L* is determined through the fixed input proportion.

The economic and ecological effects of an expansion of the protected area is found by taking the total differential of *(11)* and *(1)* and *(2)* (with $dX/dt = dY/dt = 0$) (for details, see Appendix 2). With a non-binding constraint on land use, the state gazettes non-cultivated land when expanding the protected area[13]. This means that more habitat protection displaces the local people from pre-hunting areas without restricting their rights to utilize land in agricultural production. The effect on hunting effort and land use is unclear. However, equation *(11)* demonstrates that an expansion of the protected area increases the stock density in the outer area. This gives more dispersal into the protected area and leads to a more dense population here. Because of increased stock densities, there must be a positive effect on the aggregate stock of gazetting non-cultivated land for wildlife protection. The conclusion is therefore that more protection gives more wildlife even if the local people increase their hunting effort.

The next step is to investigate how this intervention affects the economic conditions of the local people. Recall from fishery economics that Sanchirico and Wilen (2001) claim that a marine reserve may benefit the fishermen through increased aggregate catch, if increased dispersal from the marine reserve compensates for the foregone harvest in the reserve. In the present model, however, the effect on the wildlife harvest is not an adequate measure of the impact on the economic conditions of the local people. Instead we need to investigate the effect on the net income in *(7)* in optimum. This is done by taking the differential of *(7)* with respect to w, when accounting for the effect working through a changing stock density (see Appendix 2). In contrast to marine reserves, it turns out that there is no potential for improved human welfare of expanding a protected area. There is a direct negative effect working through the reduced income from hunting due to restricted hunting rights. While the expansion of the protected area increases the stock density in the outer area, this cannot offset the direct negative effect. This means that positive effect on the wildlife density cannot compensate the local people for the foregone hunting ground. This negative income effect is strengthened by the fact that a more dense wildlife population imposes further damage to agricultural crops. Gazetting non-cultivated land for habitat protection will therefore promote wildlife conservation at the expense of human welfare. See also Table 1 in section 4.2.

[13] In addition, an expansion of the protected area is feasible through expropriation of agricultural land in the outer area. In this case, the local people are displaced from both pre-hunting areas and pre-cultivated land. However, because land is homogeneous and there is no investment cost in land, the local people simply move their agricultural production to the pre non-cultivated areas. Therefore, as long as the constraint on land use is non-binding, the conservation-effect of gazetting non-cultivated or pre-cultivated land is identical.

4.2 The constraint on land use is binding

Assume that land is a scarce factor to the local people living in the outer area. This is the case if, relatively speaking, the protected area is widespread, the agricultural productivity is high, and/or the marginal wildlife-induced damage to crops is low. In such a scenario, the local people settle in a corner solution for cultivated land, i.e. $L = (1 - w)$ and $\lambda > 0$ from *(10)*. Hence, the marginal return on land cultivation exceeds the marginal damage in *(9)*. See also Figure 2. Inserting $L = (1 - w)$ in *(8)* gives

$$(12) \qquad P_h f(1 - w)Y = P_A \mu\alpha(1 - E_h)^{\alpha - 1}(1 - w)^{1 - \alpha}$$

Equation *(12)* states that the local people will divert effort to hunting until the marginal benefit of hunting equals the marginal cost. The marginal cost reflects the alternative cost of hunting, namely the foregone return on agricultural production. In contrast to the non-binding scenario, the hunting effort E_h is now determined from the economic first order condition for a given wildlife density *Y*. This is because the amount of cultivated land *L* is fixed at $(1 - w)$ and, hence, the hunting effort follows from the fixed input proportion $(1 - E_h)/(1 - w) = [P_A \mu\alpha / (P_h f(1 - w)Y)]^{1/(1-\alpha)}$.

The economic equilibrium for a given wildlife density in the outer area is illustrated in Figure 3. Here, the marginal benefit from hunting (MBH) is measured along the left-hand vertical axis, while the marginal benefit from agricultural production (MBA) is measured along the right-hand axis. The optimal hunting effort is determined by the intersection between the two curves. Equation *(12)* shows that an expansion of the protected area, i.e. an increase in w, has a direct negative effect on the marginal return on labour in agriculture. This is because the State must expropriate cultivated land in order to expand the protected area and this is new compared to the non-binding case. Consequently, the MBA curve in Figure 3 shifts down, which works in the direction of increased hunting effort. However, restricted hunting rights reduce the marginal return on hunting, which shifts the MBH curve down. This leads the local people to direct less effort towards hunting. The total effect on hunting effort is therefore unclear. If restricted hunting rights affect the local people less than reduced cultivated land, i.e. $\left|\partial^2 H / \partial E_h \partial w\right| < \left|\partial^2 A / \partial(1 - E_h)\partial w\right|$, they will reply to habitat protection by directing more effort to hunting. This is illustrated in Figure 3 by a stronger downward shift in the MBA curve.

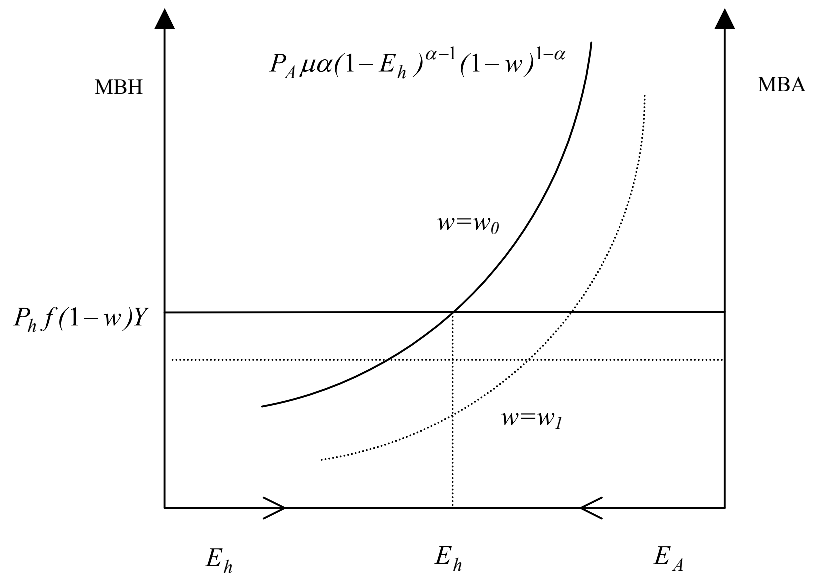


Figure 3: The maximum condition for the hunting effort E_h.
The constraint on land is binding.
Y is fixed, $< w_1$.

If the harvesting effort changes, however, both the wildlife densities and the aggregate stock will change, since they all depend on E_h. The first order condition in *(12)* and the ecological equilibrium in *(1)* and *(2)* (with $dX / dt = dY / dt = 0$), determine simultaneously the optimal hunting effort and the stock densities. Again, the aggregate stock follows from $dS / dt = 0$. Differentiation of these equations with respect to w gives the impact of a protected area expansion (for details, see Appendix 2). In contrast to section 4.1, it turns out that the effect on wildlife conservation is ambiguous. The mechanism works as follows. Consider first the direct effect. Because more animals are protected from hunting for a given hunting effort, the aggregate stock S increases.

Second, we have the indirect effect working through the hunting decision of the local people. As discussed above, restricted user rights to wildlife reduce the marginal return from labour in hunting, while restricted user rights in agriculture reduce the marginal return from labour in crop production. These have opposite effects on the hunting effort. As argued, if the local people respond less to the closed hunting ground than the loss of cultivated land, they will divert more labour effort towards wildlife exploitation. This will be the case in areas where the local people rely heavily on agriculture as a land use so that expropriation of cultivated land represents a considerable income loss. In this case, the indirect effect on wildlife conservation implies less wildlife in the outer area and a smaller aggregate stock. The total effect on wildlife conservation is therefore unclear. Contrary to the non-binding scenario, this demonstrates that protected areas which restrict the user rights to wildlife and cultivated land may reduce the degree of wildlife conservation. This obscure result occurs because the constraint on land use in agriculture is binding, meaning that there is an alternative use of the protected land in agricultural production.

The final part of this analysis is to investigate how expropriation of cultivated land affects the economic conditions of the local people living with wildlife. Again, differentiation of *(7)* with respect to w, when taking into account the effect working via a changing wildlife stock, gives the effect on local income in optimum. There are three possible outcomes regarding wildlife conservation and local welfare, and these are summarised in the second column of Table 1. Assume first that an expansion of the protected area fails and results in a smaller degree of wildlife conservation. As reported in the table, this results in poorer economic conditions for the local people. Therefore, this model predicts that where protected areas have failed in promoting wildlife conservation, they have also caused a degradation of human welfare[14].

[14] Obviously, protected areas cannot promote local welfare at the expense of wildlife conservation. In the case of a binding constraint on land use and $dS/dw>0$, $d\pi/dw>0$ must indicate that the local people were utilizing 'too much' land for agricultural production prior to the expansion of the protected area. In this case, profit-maximization requires that the local people choose an interior solution for cultivated land (i.e. a non-binding constraint on land use).

Table 1: The welfare effect of an increase in w in equilibrium.

	Non-binding constraint on land use	Binding constraint on land use*		
S	+	÷	+**	
π	÷	÷	÷ z low or z high; c high, P_h low	+ z high; c low, P_h high

* The welfare effect is conditioned by the impact on wildlife conservation.
** Here, it is assumed that the wildlife stock in the outer area increases, $d(1-w)Y/dw > 0$. The magnitude of dY/dw increases with z. z 'high' is interpreted as the case where increased dispersal from the protected area compensates for the foregone hunting ground. z 'low' means that dispersal from the protected area cannot compensate for the foregone hunting ground.

An expansion of the protected area promotes wildlife conservation if the local people respond to restricted user rights by devoting more effort to agricultural production. As shown in Table 1, the resulting effect on local welfare is ambiguous and dependent on the moving rate of wildlife and the benefit and cost of living with wildlife. First, the income from hunting is reduced if increased dispersal from the protected area cannot compensate for the foregone return from the pre-hunting ground. This will be the case if the moving rate z is 'low'. Then, an expansion of the protected area promotes wildlife conservation at the expense of human welfare[15]. Although the systems are quite different, the same conclusion is drawn by Sanchirico and Wilen (2001). Second, the income from hunting increases if increased dispersal from the protected area exceeds the foregone return from the pre-hunting ground. As for marine reserves, this requires a 'high' moving rate z. Still, and in contrast to marine reserves, there is a negative impact on human welfare as more animals in the outer area cause more damage to agricultural crops. If the cost of living with wildlife is sufficient above the benefit (i.e. c 'high' and P_h 'low'), wildlife conservation is promoted at the expense of human welfare. On the other hand, the welfare effect is positive if the benefit of living with wildlife exceeds the cost.

[15] Here, the effect on local welfare stems from the assumption that more wildlife in the entire area is a result of higher densities and stock sizes in both sub-areas. See Table 1.

The results summarised in Table 1 demonstrate that there is a potential for a double payoff to emerge where both wildlife and the local people benefit from an expropriation of cultivated land. On the other hand, while the degree of wildlife conservation increases, the welfare of the local people reduces when the State gazettes non-cultivated land. The reason for these adverse effects on human welfare lies in the assumed constant return to scale in the agricultural production function in *(5)*. In presence of this formulation, the equilibrium stock density in the outer area is determined by the size of the protected area and the fixed economic parameter values when the constraint on agricultural land is non-binding. Hence, in this case, the stock density in the outer area is independent of the hunting effort and the stock density in the protected area. The implication of this is that the welfare of the local people in optimum is only influenced by the direct effect of a changing w and the following effect working through a changing Y. That is, the welfare in optimum is independent of how a changing w affects E_h and X and, thereby, the dispersal from the protected area. The mechanism at work is quite different when the constraint on agricultural land is binding. Then, the equilibrium stock density in the outer area is affected by the hunting effort, the density in the protected area and the dispersal from this area onto the open land. In presence of this interdependency in equilibrium, the local people will benefit from any increased dispersal from the protected area. Precisely this effect gives the potential for a double payoff to emerge from expropriation of cultivated land.

5. Discussion and concluding remarks

Establishing national parks and other types of protected areas have been the traditional approach to natural resource conservation in Africa. However, protected areas have during the past decade been viewed as having failed to preserve wildlife on that continent (Kiss 1990, Barrett and Arcese 1995, Gibson and Marks 1995). Martin (1993) explains this failure by claiming that the budgets and funds are too small to finance the activities necessary to combat illegal hunting on protected land. This paper gives an alternative explanation of what may have caused the failure of protected areas. In contrast to Martin, the analysis demonstrates that protected areas may cause wildlife degradation, even when anti-poaching law enforcement succeeds in eliminating illegal hunting in the gazetted area.

What has been analysed in this paper is a wildlife management system where land is gazetted as a protected area in order to conserve wildlife. The ecosystem of consideration is of fixed size and consists of two sub-areas – the protected area and the outer area – over which the wildlife stock disperses. The outer area is settled by humans who utilize this area for wildlife hunting and agricultural production. The local people have user rights to wildlife and land for cultivation in the outer area, but they do not have the property rights. Related to the land use in the outer area, this paper distinguishes between two ways of gazetting land. First, the state gazettes non-cultivated land. This policy restricts the local people's user rights to wildlife by withdrawing former hunting grounds without interfering with their rights to cultivate land. Technically, this is the case where the constraint on land use is non-binding. Second, the state expropriates cultivated land, a policy which restricts the local people's user rights to both wildlife and land for cultivation. In this scenario, the constraint on land use is binding.

The main point of the analysis is to find out under which conditions protected areas may fail in conserving wildlife. In addition, the analysis focuses on the economic impact of protected areas by investigating the effect on human welfare. It is shown that the actual outcome of habitat protection depends critically on whether the constraint on land use is binding. Only when the constraint is non-binding will protected areas with certainty increase the wildlife stock. This scenario is quite similar to a marine reserve creation with no alternative use of the marine habitat. However, in contrast to marine reserves, there is no potential for improved human welfare of gazetting non-cultivated land.

Protected areas work quite differently from marine reserves when the constraint on land use is binding and the State expropriates cultivated land for wildlife protection. This discrepancy stems from the alternative use of protected land as land for agricultural production. The model demonstrates that an expansion of the protected area causes a degradation of wildlife if the impact of lost cultivated land is high relative to the impact of lost hunting grounds. If this is the case, the local people will compensate themselves by devoting more time on hunting. In contrast, if the impact of less cultivated land is low relative to the impact of restricted hunting grounds, the local people respond to land expropriation by devoting more time on agricultural production and, thereby, reduce the time spent hunting. In this case, an expansion of the protected area promotes wildlife conservation. Then, a double payoff will emerge if the wildlife-induced damage to agricultural crops is small and the increased dispersal from the protected area compensates for the foregone wildlife harvest on the pre-hunting grounds.

It is important to note, however, that this model simplifies the interaction between the wildlife population dynamics and the human activities. In the ecological part of the model, the quality of land as habitat for wildlife is considered constant and independent of the agricultural use in the outer area. In reality, however, unexploited areas may generate more wildlife than cultivated land. Therefore, the analysis overlooks a positive effect on wildlife conservation as protected areas displace agricultural activities in the wildlife habitat. However, the weakness of omitting this connection becomes less apparent as there is assumed to be no cost in converting land to agricultural use in the economic part of the model. In reality, investing in land is costly and time consuming. This strengthens the negative impact on the local people of an expansion of the protected area and may therefore lead to wildlife degradation. Implementing investment costs may even cause failure when the constraint on land use is non-binding.

It is also important to notice the lack of economic compensation to the local people of restricted user rights to wildlife and land. With the growing recognition of the failure of protected areas, international conservation organizations and African governments have developed a new approach to wildlife conservation, namely the Integrated Conservation and Development Project (ICDP) (see Kiss 1990, Barbier 1992, Wells and Brandon 1992, Barrett and Arcese 1995, Barrett and Arcese 1998, Songorwa 1999). The central issue in ICDPs is benefit sharing, or compensation for restricted user rights, for instance through income transfers from the tourism sector. Neglecting such compensation schemes may suppress a potential positive effect on local welfare of protected areas. Still, because there is no broad-based evidence in the literature suggesting that existing

ICDPs can fully compensate the local people for the loss of user rights, such benefit-sharing schemes have not been taken into account in this paper (see also Barrett and Arcese 1995, Gibson and Marks 1995, Emerton 1998, Songorwa 1999).

References

Armstrong, C.W. and S. Reithe (2001): "Comment: Marine reserves – will they accomplish more with management costs?", *Marine Resource Economics*, 16(2), 165-175.

Barbier, E.B. (1992): "Community-based development in Africa". In Swanson, T.M. and E.B. Barbier (eds.): *Economics for the Wilds: Wildlife, Wildlands, Diversity and Development*, Earthscan, London, 103-135.

Barrett, C.B. and P. Arcese (1995): "Are integrated conservation-development projects (ICDPs) sustainable? On the conservation of large mammals in sub-Saharan Africa", *World Development*, 23(7), 1073-1084.

Barrett, C.B. and P. Arcese (1998): "Wildlife harvest in integrated conservation and development projects: Linking harvest to household demand, agricultural production, and environmental shocks in the Serengeti", *Land Economics*, 74(4), 449- 465.

Bromley, D. (1991): *Environment and Economy*, Blackwell, Oxford.

Bulte, E.H. and R.D. Horan (2000): "Habitat conservation, wildlife extraction and agricultural expansion", forthcoming in *Journal of Environmental Economics and Management*.

Bulte, E.H. and D. van Soest (1999): "A note on soil depth, failing markets and agricultural pricing", *Journal of Development Economics*, 58(1), 245-254.

Caughley, G. and A.R.E. Sinclair (1994): *Wildlife Ecology and Management*, Blackwell Science, Cambridge.

Conrad, J.M. (1998): "The bioeconomics of marine sanctuaries", *Journal of Bioeconomics*, 1(2), 205-217.

Dixon, J.A. and P.B. Sherman (1991): *Economics of Protected Areas. A new look at Benefits and Costs*, Earthscan Publications Ltd., London.

Emerton, L. (1998): "Why wildlife conservation has not economically benefited the communities in Africa", *Community Conservation Research in Africa*, Paper no 5, Institute for Development Policy and Management, University of Manchester, Manchester.

Gibson, C.C. and S.A. Marks (1995): "Transforming rural hunters into conservationists: An assessment of community-based wildlife management programs in Africa", *World Development*, 23(6), 941-957.

Hannesson, R. (1998): "Marine Reserves: What Would They Accomplish? ", *Marine Resource Economics*, 13(3), 159-170.

Hayami, Y. and V.W. Ruttan (1985): *Agricultural Development. An International Perspective*, The Johns Hopkins University Press, London.

Huffaker, R.G., M.G. Bhat and S.M. Lenhart (1992): "Optimal trapping strategies for diffusing nuisance-beaver populations", *Natural Resource Modelling*, 6(1), 71-97.

Kiss, A. (1990): "Living with wildlife: Wildlife resource management with local participation in Africa", *Technical Paper*, 130, World Bank.

Leader-Williams, N. and S.D. Albon (1988): "Allocation of resources for conservation", *Nature*, 336 (December), 533-535.

Lopez, R. (1998): "Agricultural intensification, common property resources and the farm-household", *Environmental and Resource Economics*, 11(3-4), 443-458.

Marks, S.A. (1984): *The Imperial Lion: Human Dimensions of Wildlife Management in Africa*. Westview Press, Boulder.

Martin, R.B. (1993): "Should wildlife pay its way? ", Working Paper, Department of National Parks and Wildlife Management, Zimbabwe.

Norton-Griffiths, M. (2000): "Wildlife losses in Kenya: an analysis of conservation policy", *Natural Resource Modeling*, 13(1), 13-34.

Pezzey, J., C.M. Roberts and B.T. Urdal (1998): "A simple bioeconomic model of a marine reserve", *Ecological Economics*, 33(1), 77-91.

Pulliam, H.R. (1988): "Sources, sinks, and population regulation", *The American Naturalist*, 132, 652-661.

Sanchirico, J.N. and J.E. Wilen (2001): "A bioeconomic model of marine reserve creation", *Journal of Environmental Economics and Management*, 42(3), 257-276.

Skonhoft, A. (1999): "Conservation of wildlife. A bioeceonomic model of a wildlife reserve under the pressure of habitat destruction and harvesting outside the reserve", Working Paper, Norwegian University of Science and Technology.

Skonhoft, A. (1998): "On the optimal exploitation of terrestrial animal species", *Environmental and Resource Economics*, 13(1), 45-57.

Skonhoft, A. and J.T. Solstad (1998): "The political economy of wildlife exploitation", *Land Economics*, 74(1), 16-31.

Smith, V.L. (1975): "The primitive hunter culture, pleistocene extinction, and the rise of agriculture", *Journal of Political Economy*, 83(4), 727-755.

Songorwa, A.N. (1999): "Community-based wildlife management (CWM) in Tanzania: Are the communities interested? ", *World Development*, 27(12), 2061-2079.

Swanson, T.M. and E.B. Barbier (1992): *Economics for the Wilds: Wildlife, Wildlands, Diversity and Development*, Earthscan, London.

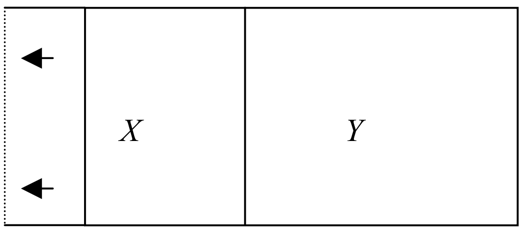
Wells, M. (1992): "Biodiversity conservation, affluence and poverty: mismatched costs and benefits and efforts to remedy them", *Ambio*, 21(3), 237-243.

Wells, M. and K. Brandon (1992): *People and Parks: Linking Protected Area Management with Local Communities*, World Bank, World Wildlife Fund and U.S. Agency for International Development, Washington DC.

Wright, R. (1999): "Wildlife management in the national parks: questions is search of answers", *Ecological Applications*, 9, 30-36.

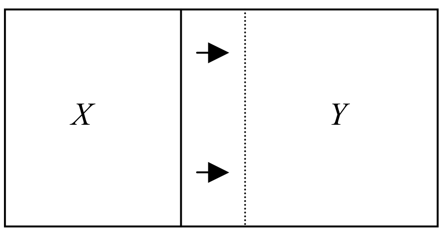
Appendix 1

Migration between a protected area (or marine reserve) and an outer area is also of focus in Conrad (1998), Skonhoft (1999), and Sanchirico and Wilen (2001). The point of departure for these analyses is that the migration is density dependent, i.e. the net in/out migration of the protected area depends on the ratio of the stock size to the carrying capacity within each sub-area. This means that increased carrying capacity in one sub-area, which reduces its stock density, leads to reduced migration out of this particular area. Skonhoft (1999) interprets the carrying capacity of each sub-area as proportional to the size of the respective area (see also Pezzey et al. 2000). In addition, the size of the ecosystem is not fixed, meaning that it is possible to increase the size of one sub-area without altering the size of the other area. This is illustrated in the figure below, where the size of the protected area increases as indicated by the left-pointing arrows. In this system, more protected land means more space for the animals, which has a direct negative effect on the stock density in the protected area. Consequently, the migration into the protected area increases due to a relatively low density here.



Protected area Outer area

In the present analysis, the ecosystem is of a fixed size. This means that an increment in the protected area takes place by altering the (non-physical) border between the park and the outer area. Consequently, more protected land is followed by an equal reduction of the open area, as illustrated by the right-pointing arrows in the figure below.



Protected area Outer area

In this system, the effect on wildlife migration of altering the size of the protected area is somehow different from Skonhoft's model (1999). Here, there is no direct effect on the stock densities. Instead, expanding the protected area reduces the probability of an animal in the pre-protected area moving into the outer area. Recall that this probability is given by $(1-w)$. For a given stock density *X*, this effect works in the direction of reduced migration out of the protected area. At the same time, for a given stock density *Y*, the probability *w* that an animal in the pre-outer area is included in the protected area increases. However, when the outer area shrinks, its stock size $(1-w)Y$ is reduced for a given *Y*. Translated into a changing stock density in the protected area (i.e. $zw(1-w)Y/w = z(1-w)Y$, see the main text), these latter effects work in the direction of reduced migration into the protected area. A fixed positive harvesting rate in the outer area gives $X > Y$ (see main text), and, consequently, the total effect on net migration into the protected area is positive.

Both Conrad (1998) and Skonhoft (1999) consider the rate of change in biomass *x* and *y* in two areas with different carrying capacities K_x and K_y and with a dispersal function given as $s(x/K_x - y/K_y)$. Skonhoft extends this dispersal function by taking into account the fact that dispersion due to different sex and age composition of the two sub-populations can be skew. The function is therefore given by $s(\beta x/K_x - y/K_y)$. Sanchirico and Wilen (2001) define the stock densities as $X = x/K_x$ and $Y = y/K_y$ and specify the dispersal function as $s(X-Y)$. In the present analysis, the carrying capacities per square kilometre equal unity so that $X = x/w$ and $Y = y/(1-w)$, β equals 1, and the migration rate *s* equals *zw(1-w)*.

Appendix 2

1. Non-binding constraint on land use

Taking the total differential of the ecological equilibrium $dX/dt = dY/dt = 0$ in *(1)* and *(2)* yields

$$\begin{bmatrix} r(1-2X)-z(1-w) & z(1-w) \\ zw & r(1-2Y)-zw-f(1-w)E_h/(1-w) \end{bmatrix}\begin{bmatrix} dX \\ dY \end{bmatrix}$$

$$(A1) \quad = \begin{bmatrix} z(Y-X) \\ z(Y-X)-E_hY\left[f'(1-w)-f(1-w)/(1-w)\right]/(1-w) \end{bmatrix} dw$$

$$+\begin{bmatrix} 0 \\ f(1-w)Y/(1-w) \end{bmatrix} dE_h$$

The determinant $D = \left[r(1-2X)-z(1-w)\right]\left[r(1-2Y)-zw-f(1-w)E_h/(1-w)\right]-z^2w(1-w)$ is positive from the condition of ecological stability. Figure 1 shows that a given positive harvesting effort shifts the Y-isocline down for a fixed positive wildlife density in the protected area. Consequently, $dY/dE_h\big|_{dX=0} = (f(1-w)Y/(1-w))/\left[r(1-2Y)-zw-f(1-w)E_h/(1-w)\right] < 0$, meaning that the denominator is negative. The system settles in a new ecological equilibrium with reduced densities. It follows from *(A1)* that $dY/dE_h = \left[r(1-2X)-z(1-w)\right]f(1-w)Y/D(1-w) < 0$. The differential of *(11)* together with *(A1)* gives the comparative static results in the case of a non-binding constraint on land use as

$$\begin{bmatrix} r(1-2X)-z(1-w) & 0 \\ zw & -f(1-w)Y/(1-w) \end{bmatrix}\begin{bmatrix} dX \\ dE_h \end{bmatrix}$$

$$(A2) \quad = \begin{bmatrix} z(Y-X)-z(1-w)\alpha f'(1-w)Y/f(1-w) \\ \rho \end{bmatrix} dw$$

where the determinant of the system $-\left[r(1-2X)-z(1-w)\right]f(1-w)Y/(1-w)$ is positive.

The sign of $\rho=-\left[r(1-2Y)-zw-f(1-w)E_h/(1-w)\right]\alpha f'(1-w)Y/f(1-w)$ $+z(Y-X)-\left[f'(1-w)-f(1-w)/(1-w)\right]E_hY/(1-w)$ is unclear. The corresponding change in the aggregate stock density equals $dS=wdX+(1-w)dY-(Y-X)dw$ where dY is given from the differentiation of *(11)* and dX is given from *(A2)*.

The input proportion in agricultural production is found by inserting *(11)* in *(8)* (or *(9)*) and equals $(1-E_h)/L=P_A\alpha c/\left[P_hf(1-w)(1-\alpha)\right]$. Differentiation of this with respect to L, E_h, and w gives

(A3) $$dL=-\left[L/(1-E_h)\right]dE_h-\left[Lf'(1-w)/f(1-w)\right]dw$$

Taking the differentiation of *(7)* with respect to w, and taking into account the effect working through Y, gives the effect on local welfare of expanding the protected area.

(A4) $$\partial\pi/\partial w=P_hE_h\left[f(1-w)dY/dw-f'(1-w)Y\right]-P_AcLdY/dw$$

Here, the term $f(1-w)dY/dw$ reflects that the income from hunting increases due to increased stock density, while $f'(1-w)Y$ reflects reduced income from hunting due to the foregone return from the pre-hunting ground. It is easy to show that the net effect on the income of the local people is negative. Inserting for $dY/dw=\alpha f'(1-w)Y/f(1-w)>0$ from the differentiation of *(11)* with respect to w gives $d\pi/dw=-f'(1-w)Y\left[P_hE_h(1-\alpha)+\alpha P_AcL/f(1-w)\right]<0$.

2. Binding constraint on land use

Recall that $dS=wdX+(1-w)dY-(Y-X)dw$, where dX and dY are given from *(A1)*.

With a binding constraint on land use, the comparative static results are derived from differentiation of this and *(12)* with respect to w

$$\textit{(A5)}\quad \begin{bmatrix} 0 & \delta \\ 1 & \sigma \end{bmatrix} \begin{bmatrix} dS \\ dE_h \end{bmatrix} = \begin{bmatrix} \tau \\ \theta \end{bmatrix} dw$$

The sign of

$\delta = -P_A\alpha(1-\alpha)\mu(1-E_h)^{\alpha-2}(1-w)^{1-\alpha} + P_h f(1-w)^2 Y[r(1-2X) - z(1-w)]/D(1-w)$

is negative so that the determinant is positive. The sign of

$\sigma = -f(1-w)Y[r(1-2X) - z(1-w) - zw]/D$ is positive, while the signs of

$\theta = (X-Y) - [z(1-w)(X-Y) + E_h Y(f'(1-w) - f(1-w)/(1-w))][r(1-2X) - z(1-w) - zw]/D$
$- zw(X-Y)[r(1-2Y) - zw - z(1-w) - f(1-w)E_h/(1-w)]/D$ and

$\tau = P_h f'(1-w)Y - P_A\alpha(1-\alpha)\mu(1-E_h)^{\alpha-1}(1-w)^{-\alpha} + P_h f(1-w)z^2 w(Y-X)/D$

$-[r(1-2X) - z(1-w)][z(Y-X) - E_h Y(f'(1-w) - f(1-w)/(1-w))]P_h f(1-w)/D$

are unclear.

Again, differentiation of *(7)* with respect to w, and taking into account the effect working through Y, gives the effect on local welfare of expanding the protected area.

$$\textit{(A6)}\quad \begin{aligned} \partial\pi/\partial w &= P_h E_h[f(1-w)dY/dw - f'(1-w)Y] - P_A\mu(1-\alpha)(1-E_h)^{\alpha}(1-w)^{-\alpha} \\ &\quad - P_A c[(1-w)dY/dw - Y] \end{aligned}$$

Here, dY/dw is determined in *(A1)* when accounting for the effect working through the hunting effort in *(12)*.

Chapter 3

Property rights and natural resource conservation. A bio-economic model with numerical illustrations from the Serengeti-Mara ecosystem

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Abstract

This study develops a model for wildlife species migrating seasonally between a conservation area and a neighbouring area. When being outside the conservation area, harvesting takes place by a group of peasants. The local people have two motives for harvesting; to get rid of 'problem' animals as roaming wildlife destroys crops and agricultural products, and hunting for meat and trophies. Depending on the specification of the property rights, the harvesting is legal or illegal. It is demonstrated that it is far from clear which of the two property rights regimes that gives the highest wildlife abundance. Hence, contrary to what is argued for in the literature, handing the property rights over to the local people means not automatically more wildlife and a more 'sustainable' resource utilization. The reason lies in the nuisance motive for harvesting. The exploitation under the two different property rights regimes are illustrated by numerical calculations with data that fits reasonable well with the exploitation of the wildebeest population in the Serengeti-Mara ecosystem.

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1. Introduction

For a long time it has been recognised that institutions play an important role in natural resource management, and that the specification and function of property rights to a large extent determine whether resources can be utilized in a sustainable way. These dimensions will be at the focus in the present study when analysing the management and exploitation of wildlife in a sub-Saharan Africa context with example from the Serengeti-Mara ecosystem. When considering natural resources in the form of wildlife in this region, as in other regions in the sub-Saharan Africa, central issues are the behaviour of the rural people living close to the wildlife, and the interaction between the rural people, the wildlife and the agency managing and having the legal property rights of the wildlife. Often this interaction represents conflicting interests; both the legal owner of the wildlife (usually the State, or large private landowners) and the rural people claim their rights to reap the benefits of the huge amount of wildlife resources. In addition, the costs of having abundant wildlife populations differ between them. These conflicts, rooted in the prevailing property structure and its functioning, have serious implications for the resource exploitation and, thus, on the management of the wildlife in general (Marks 1984, Kiss 1990, Swanson and Barbier 1992, Naughton-Treves and Sanderson 1995, Sinclair and Arcese 1995, Skonhoft and Solstad 1998, Bulte and van Kooten 1999).

The common perception in the literature is that local communities will support wildlife conservation and reduce the wildlife offtake if they are ensured a sufficient share of the benefits from wildlife (see Kiss 1990, Swanson and Barbier 1992, Mangel et al. 1996). Under what conditions a community based management system results in a higher wildlife abundance and more conservation than the polar scheme where the local people have no legal rights to wildlife exploitation, is analysed in the present paper. The starting point is that we have a protected area, a national park or a conservation area of fixed size, with no harvesting of the wildlife. The protected area is the basic living area of the species, but they roam freely in and out of the park. When being outside, the game destroys the crops of the peasants living in the vicinity of the park and hence, the wildlife represents a nuisance for the local people. The park agency has the property rights of the wildlife within as well as outside the protected area, but illegal harvesting takes place outside as the property rights is not effectively protected here. This is the first regime where the local people have no property rights[1]. In the next step, the property rights are handed over to the local people and hence, the exploitation of the wildlife outside the protected area takes place in a legal manner[2]. The different degree of wildlife utilization is compared under these two regimes, and contrary to what is argued for in the literature the first property rights scheme may result in the highest wildlife abundance. Analysing the driving forces behind such an outcome is the main contribution of this paper.

The hunting activity and the utilization of the wildlife under these two regimes are exemplified by the migration of wild herbivores in the Serengeti-Mara ecosystem. The migration pattern here, which is seasonal and related to rainfall and food supply, is also used as a motivation for the ecological part of the model. Parts of the year the wildlife is staying within the huge Serengeti national park and it is assumed that no human extraction takes place here[3]. However, when being outside there is hunting. The conceptual framework for analysing the exploitation of terrestrial species when there is migration, and hunting is a seasonal activity, is another contribution of the paper.

[1] Economists frequently confuse what is meant with property rights. Daniel Bromley understands property as a benefit stream and a property right as 'the capacity to call upon the collective to stand behind one's claim to a benefit stream' (Bromley 1991, p.15). Possession of inviolable property rights presupposes that the rights are authorised by law and that the law is effectively enforced by the state. Hence, a well-functioning property regime is characterised by, firstly, legally *welldefined* property rights and, secondly, effectively *protected* property rights. The existence or non-existence of these two factors defining the functioning of the property structure can be used to classify three different types of regimes. First of all we have the case where there is legally well-defined ownership and perfect state protection (exclusive rights, perfect law enforcement). This is in accordance with the classical Clark (1973) model. Secondly, we have the case where the ownership is legally well defined but not adequately enforced by the government. Under such a scheme, often due to lack of societal recognition of the property rights in place, the management is likely to be affected by conflicting property rights claims. Finally, we have the case where there is no legally defined ownership; that is, the open-access case regime (Gordon 1954). Hence, the second of these regimes is the starting point in the following analysis.

[2] These schemes are polar because the local people obviously can obtain other rights than full property rights of the wildlife when being outside the park, say, through a specific harvesting quota, the rights to harvest problem animals, and so forth (cf. the concluding section).

[3] In reality, poaching takes place within as well as outside the borders of Serengeti National Park (see Sinclair and Arcese 1995). However, to simplify the theoretical framework, without altering the qualitative results, we assume that hunting is only taking place in the outer area.

To simplify the following analysis we consider only two areas; the protected area and the whole boundary region collapsed into one boundary area. The protected area is the basic living area of the wildlife and is owned and managed by the State. No human extraction is allowed here, and hence, non-consumptive benefit (tourism, existence value, etc.) is the only benefit of this area. However, the park manager play no role in the subsequent model as the management of this area is supposed to have no stock effects (no harvesting, no culling)[4]. The peasants are living in the boundary area, where the wildlife roams during the migration season. They are involved in two production activities; agricultural crop production and wildlife hunting (see, e.g., Barrett and Arcese 1998). First, we study today's situation where the local people have no property rights, and hence, they hunt illegal. This is in accordance with the 'open-access' model of Smith (1975). Under this scenario we disregard any enforcement use and consequently, the park manager is passively benefiting from non-consumptive utilization of wildlife in the protected area. Because of small funds and large areas, this represents a good approximation to the present reality (see also footnote 1). In a next step, we study what happens when the property rights over the wildlife are given to the local people.

We start by formulating the ecological model and the migration pattern in section 2. In section 3 the benefit function of the local people is presented, and in section 4 it is studied how this translates into harvesting when they have no property rights. In this section, as later, it will for simplicity be assumed that only the current benefit, or current utility, is steering the harvesting activity. In section 5 the model is analysed when the local people have the property rights over the wildlife. In section 6 the two different property rights regimes are illustrated by numerical calculations with data that fits reasonable well to the exploitation of the wildebeest population in the Serengeti-Mara ecosystem.

[4] For a critical review of this assumption, see, e.g., Wright (1999).

2. The ecology

As already noted, we consider species migrating over the year in a well defined way, and where harvesting only takes place when the species are outside their core area, i.e., the conservation area or the national park. Because the seasonal migration is related to rainfall and food supply, there is no density-dependent factors influencing the species flow. For simplicity it is assumed that the whole wildlife population is leaving the core area, and we separate sharply between the periods of the year when the wildlife is roaming outside and when the species are staying inside of the protected area. These two periods are synonymous with the 'fishery season' and 'between season' in Homans and Wilen (1997)[5]. We start by formulating the hunting season dynamics while the between season dynamics and the steady state follow next.

Migration and harvesting

As in Homans and Wilen (1997) (see also Getz and Haight 1989), we suppose that the hunting effort of the local people is utilized at a constant rate throughout the period when hunting takes place, i.e., when the wildlife is roaming outside the conservation area. The wildlife harvesting function is specified as a Schaefer function giving the harvest at time *t* as

$$(1) \qquad h(t) = qaX(t),$$

where $X(t)$ is the wildlife stock (measured as biomass), while a is the fixed harvesting effort. q is a parameter, the so-called catchability coefficient.

When being within the conservation area the wildlife grows in a density-dependent fashion (see below), and when being outside natural mortality is ignored[6]. Outside the conservation area the stock therefore shrinks according to $dX(t)/dt = -qaX(t)$. The stock level outside the conservation area at time *t* is accordingly

[5] There are few, if any, papers analysing time discrete models with migration for terrestrial animal species within a bio-economic concept. Fancy et al. (1994) formulate a time discrete migration model for the porcupine caribou herd. The migration here is also seasonal, but this paper has no economic content.

(2) $X(t) = X_0 e^{-qat}$

where X_0 is the stock level when the species starts roaming outside the protected area.

The length of the migration period, i.e., the time when being outside the conservation area, is fixed as *T*. Hence, the stock level when the wildlife is returning back to the park is

(3) $X_T = X_0 e^{-qaT} = X_0 s$

where $0 < s = e^{-qaT} \leq 1$ is the survival rate of the species when being outside the conservation area. Under these assumptions, total amount of wildlife harvested is

(4) $H = X_0 - X_T = X_0(1 - s)$.

Natural growth and ecological equilibrium

So far we have studied a single period migration, where the stock size X_0 at the beginning of the period of migration is predetermined. However, as the number of animals harvested depends on the initial stock, the harvest changes when X_0 changes. The factors affecting the initial stock size are found by analysing the dynamics between the seasons of migration. We assume that the conservation area is the basic living area where calving takes place, meaning that the stock size returning back determines the natural growth. When $X_{T,t}$ is the stock returning back to the conservation area at time *t*, the density-dependent growth is accordingly determined as $F(X_{T,t})$. We represent natural growth by a logistic function with *K* as the carrying capacity and *r* as the maximum specific growth rate. Hence, if $X_{0,t+1}$ is the stock level at the beginning of the next period of migration, the dynamics between the migration seasons is given by

[6] Introducing natural mortality makes the model quite more complicated, but it will not change the qualitative

(5) $$X_{0,t+1} = X_{T,t} + F(X_{T,t}) = X_{T,t} + rX_{T,t}(1 - X_{T,t} / K).$$

Equation *(5)* together with the balance equation *(4)* written as $H_{t+1} = X_{0,t+1} - X_{T,t+1}$, yields $H_{t+1} + X_{T,t+1} = X_{T,t} + F(X_{T,t})$. Ecological equilibrium is defined by a constant wildlife stock returning back to the park area, $X_{T,t} = X_{T,t+1} = X_T$ [7]. In steady-state, natural growth, taking place within the conservation area, is therefore equal to harvesting, taking place in the surrounding area, $F(X_T) = H$. When substituting for the logistic growth function and equation *(3)* and using *(4)*, we obtain $rX_0 s(1 - X_0 s / K) = X_0(1 - s)$. Consequently, the ecological equilibrium is given by $X_0 = 0$ and

(6) $$X_0 = (K / rs)(1 + r - 1 / s)$$

with $(1 + r - 1/s) > 0$. From this equation it follows that the equilibrium stock returning back to the conservation area is equal to $X_T = (K / r)(1 + r - 1 / s)$. Because more harvesting effort reduces the survival rate *s*, more effort also clearly reduces the stock size, $\partial X_T / \partial a < 0$. From the equilibrium condition $X_0 = X_T + F(X_T)$, we also have $\partial X_0 / \partial a < 0$ as long as $-1 < F'$. When evaluating F' at the equilibrium, or differentiating equation *(6)* directly, this condition also writes $\left[2 / s - (1 + r)\right] > 0$. In what follows, this is assumed to hold.

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content of the analysis.

[7] Combining equation *(5)* and equation *(3)* written as $X_{T,t+1} = X_{=,t+1}s$ yields $X_{T,t+1} = s\left[X_{T,t} + F(X_{T,t})\right] = G(X_{T,t})$. It is well known (see, e.g., Clark 1990, Ch.7) that the condition for stability is that $-1 < G' < 1$ holds at the equilibrium. We have $G' = s(1 + F') = s\left[1 + (r - 2rX_{T,t} / K)\right]$ which at the equilibrium reads $s\left[2 / s - (1 + r)\right]$ when inserting for X_T (see the main text). Hence, stability demands $1 / s - (1 + r) < 0$ and $3 / s - (1 + r) > 0$. The first of these conditions is therefore the same as the condition

3. Agricultural production, crop damage, and harvesting

Above we established the ecological steady-state for a fixed amount of hunting effort. However, the hunting activity is determined by economic considerations and now these motives are outlined. Throughout we will think of the local people as a homogeneous group of peasants living in the boundary area of the park[8]. They are involved in two different production activities; agricultural production and wildlife hunting. In the model, as in reality, there are two basic motives behind hunting. In addition to the already mentioned nuisance motive where hunting takes place to get rid of 'problem' animals destroying crops and agricultural products, there is also hunting for meat or, occasionally, for trophies (Marks 1984, Barrett and Arcese 1998). As noted, because of small funds and large areas, the harvesting, when being illegal, takes place in an environment lacking any enforcement use.

We first look at the nuisance motive working through the agricultural yield function. The maximum agricultural yield, i.e., the yield without damage, depends on the amount of agricultural land, pesticides and fertiliser use, rainfall, etc., and effort use. Keeping all factors fixed except of effort, the yield function reads

$$(7) \qquad A = A(N)$$

here N is the *total* (cumulative) labour input in agricultural production over the year (seeding, harvesting and maintaining the crop). A is an increasing function of N with $A(0) = 0$, but at a decreasing rate, so that $A'(N) > 0$ and $A''(N) \leq 0$ hold.

More wildlife means more nuisance, and following Zivin et al. (2000) we assume that the damage is proportional to the amount of wildlife. Wildlife consumes a fraction β of its body weight in forage dry matter per day, and a fraction γ of this is eaten from crops.

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for obtaining an interior steady-state, while the second condition is fulfilled if $\partial X_0 / \partial a < 0$ holds (see the main text).

[8] It is therefore assumed that there are no conflicting interests among them. Hence, prevalence of individual conformity to group norms is assumed to be present. In line with traditional reasoning, it is assumed that the

The nuisance stream at time t is therefore $\gamma\beta X(t)$ with $0<\beta<1$ and $0\leq\gamma\leq 1$ as fixed constants[9]. Consequently, the cumulative damage of the wildlife roaming outside the conservation area is

$$(8) \qquad D=\gamma\beta X_0(1-s)/qa$$

when using equation *(2)* and integrating. If P_A is the crop price, assumed to be fixed, and leaving out fertiliser and pesticides costs, etc., the net crop benefit is

$$(9) \qquad B=P_A\left[A(N)-D\right],$$

with D defined in the interval $\left[0,A(N)\right]$ as the damage can never exceed the yield. The nuisance motive for harvesting is therefore working through D. More harvesting reduces the damage for a given stock size as the total amount of harvested animals increases.

In addition to the nuisance motive, we have the direct benefit of hunting in the form of meat (and trophies). The hunting benefit is given as

$$(10) \qquad V=P_H H=P_H X_0(1-s)$$

where P_H is the marginal valuation, or price, of the offtake, also assumed to be fixed.

There will be a constraint on effort used in hunting and crop growing. As a is hunting effort per unit of time and T is the length of the migration and hunting season, this resource constraint reads

elders are in charge of the group's activities (Marks 1984). This obviously means that the scenario of property rights to the local people in section 5 is implemented in its most optimistically way (cf. the concluding section).

9 We assume that hunting is the only damage control performed by the local people (see below). In addition, and in reality, damage control is also performed through fencing and other measures more directly related to protecting the crop. In the model this would have worked through γ. As we are neglecting such measures, γ is assumed to be constant.

(11) $L \geq aT + N$.

L is the total available time per year, which also can be interpreted as the total human population living in the vicinity of the conservation area. Throughout the analysis L is fixed, meaning that we are ignoring any, if possible, Malthusian mechanism. Moreover, all the time the constraint is assumed to be binding, and hence, there is always a positive opportunity cost of labour use.

When using the resource constraint, we obtain the total current benefit of the local people as

(12) $U = P_H X_0(1-s) + P_A[A(L-aT) - \gamma\beta X_0(1-s)/qa]$.

In the next section, we determine the optimal harvesting effort and wildlife population when the local people have no property rights to the wildlife roaming in the outer area. In section 5, we study the polar case when they are given full property rights.

4. No property rights of the local people

Assuming that the local people have no property rights corresponds, as already mentioned, to the stylised facts situation in sub-Saharan Africa today. When having no property rights and obtaining no legal benefit from the wildlife, they have few, if any, incentives to base their harvesting on long-term considerations. Technically, this means that they do not take the stock of wildlife into account when deciding their effort use, and hence, the number of animals entering the boundary area is treated as an exogenous variable. As already mentioned, this corresponds to what Smith (1975) calls an 'open-access' solution. See also Skonhoft and Solstad (1998).

Maximizing the benefit function *(12)* with respect to the control a, when keeping X_0 exogenous, yields the first order condition

(13) $P_A A'(N)T - P_A\gamma\beta X_0 C / a = P_H X_0 qTs$

when we have an interior solution so that effort is allocated to both production activities. C collects terms, and reads $C = (1/qa)[1-(1+qaT)s] > 0$[10]. The optimal hunting effort is therefore determined by the equality between the marginal benefit in agricultural production, corrected for damage of the roaming wildlife, and the marginal hunting benefit.

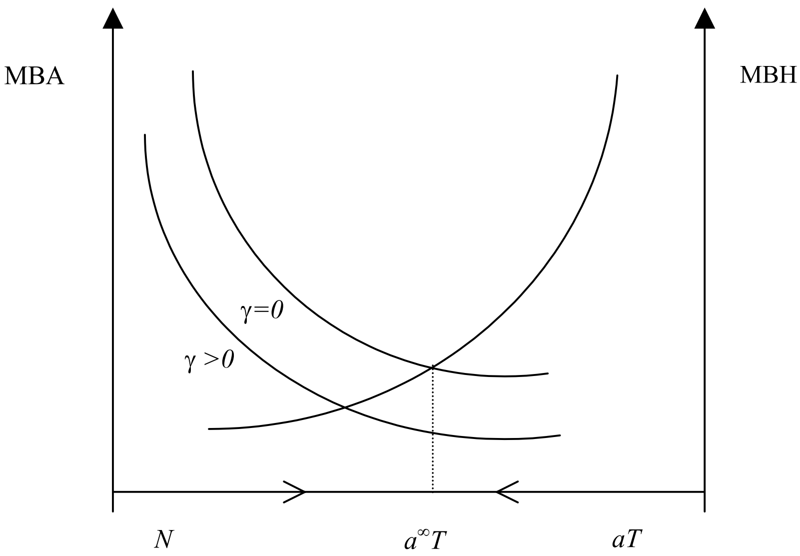


Figure 1: Economic equilibrium. No property rights regime.

Figure 1 illustrates the economic equilibrium. The marginal net benefit from agricultural production MBA is measured along the left hand vertical axis, while the marginal hunting benefit MBH is measured along the right hand axis. The second term on the left hand side of condition *(13)* reflects the nuisance motive caused by crop damage. Crop damage reduces the marginal profitability in agricultural production, and consequently, the local people channel more effort to hunting compared to a situation when crop damage is absent and $\gamma = 0$. This is illustrated by the downward shift in the marginal net agricultural benefit curve in the figure.

[10] As noticed above, the survival rate is equal to $s = e^{-qaT}$. Hence, $C > 0$ for all $T > 0$.

The economic equilibrium condition *(13)* together with the ecological equilibrium condition *(6)* defines the equilibrium values of hunting effort a^{∞} and the initial stock size X_0^{∞} (superscript '∞' denotes no property rights of the local people). In a next step, the equilibrium wildlife population returning back to the park area at the end of the season X_T^{∞} is found by using equation *(3)*. When taking the total differential of these equations, the economic and ecological forces at work can be demonstrated (for details, see Appendix 1). The results are reported in Table 1. As already indicated, a higher nuisance effect reduces the marginal productivity of agricultural production and, hence, the local people increase their harvesting effort, $\partial a^{\infty} / \partial \gamma > 0$. Consequently, the initial stock X_0^{∞} decreases, and this will also be so for X_T^{∞}. A higher price of game meat will also motivate for more hunting effort, $\partial a^{\infty} / \partial P_H > 0$, and hence, X_T^{∞} as well as X_0^{∞} decrease. A higher crop price causes the local people to divert more effort towards agriculture and the stock will increase, $\partial X_0^{\infty} / \partial P_A > 0$. This result is in line with the reasoning of Brown et al. (1993) and the analysis in Skonhoft and Solstad (1998), who find that better economic conditions in the agricultural sector always works in the direction of more wildlife conservation when the land use, as here, is fixed.

Table 1: Comparative static results

	No property rights		Property rights	
Parameter	a^{∞}	X_0^{∞}	a^{*}	X_0^{*}
γ	+	-	+	-
P_A	-	+	-/+	+/-
P_H	+	-	+/-	-/+
T	?	?	?	?
L	+	-	+	-
q	?	?	?	?
r	+	+	?	?
K	+	+	?	?

Note: When '+/-'; a reduction in a^{*} (-) is accompanied by a higher X_0^{**} (+), and vice versa. When '?'; ambiguous sign.

The effect of a higher *T* is ambiguous. First, a longer migration period reduces the productivity of hunting as an instrument of damage control. This causes the local people to channel more effort towards agricultural production. Second, the marginal product of hunting for meat increases with the period of migration, leading the local people to increase their hunting effort. Consequently, if the costs of living with wildlife exceed the benefit, then more effort is directed to agriculture. However, the effect on total hunting effort (aT) and wildlife conservation is unclear. For other results, see Table 1.

5. Property rights to the local people

Above the harvesting activity and stock sizes were found when the local people had no property rights and hence, the harvesting activity was not based on long-term considerations. Now we proceed to study what happens in the polar case when the property rights are handed over to the local people and they do no longer harvest illegally. Giving the property rights to the local people means that they are investing in wildlife and hence, take the wildlife abundance into account when allocating effort among the two production activities. Technically, the problem now is therefore to maximize *(12)* with respect to a, subject to the ecological constraint *(6)*[11].

The first order condition for this problem is

$$\text{(14)} \quad \begin{aligned} &P_A A'(N)T - P_A\gamma\beta X_0 C/a \\ &-\left[P_A\gamma\beta - P_H qa\right]qT(1-s)K\left[2/s-(1+r)\right]/qars = P_H X_0 qTs \end{aligned}$$

when still assuming an interior solution. This equation together with equation *(6)* determines the equilibrium effort a^* and equilibrium stock size X_0^*, while X_T^* again follows from equation *(3)* (superscript '*' denotes the property rights case). The difference compared to the economic equilibrium condition *(13)* in the case of no local property rights, is the third term on the left hand side. $\left[2/s-(1+r)\right]$ is here positive because $\partial X_0/\partial a < 0$ holds (see section 2). The new term, depending on the sign of

[11] A shadow price is therefore now imposed on the wildlife. It can easily be shown that it is positive only as long as $[P_A\gamma\beta - P_H qa] > 0$ holds; that is, the non-nuisance case (see the main text below). Accordingly, the shadow price is zero in the no property rights case.

$[P_A\gamma\beta - P_H qa]$, can therefore be either positive or negative, and hence, a^* can be below as well as above that of a^∞. Consequently, X_0^* can also be above or below that of X_0^∞.

The general conclusion following the model is therefore that it is unclear whether handing the property rights over to the local people will result in a higher wildlife abundance and a more 'sustainable' resource utilization. On the one hand, having the property rights and taking the size of the wildlife into account when determining hunting effort, work in the direction of a higher stock size. However, at the same time, crop damage related to the size of the wildlife will also be taken into account, and this works in the other direction. The actual outcome depends critically on whether the marginal nuisance of the wildlife stock dominates the marginal harvesting benefit; that is, the sign of the term $[P_A\gamma\beta - P_H qa]$ above. If $P_H qa > P_A\gamma\beta$ holds we therefore obtain the standard result from the literature saying that the presence of property rights means less harvesting effort and more wildlife conservation compared to the no property rights case; that is, $a^* < a^\infty$ and $X_0^* > X_0^\infty$. On the other hand, when the marginal nuisance dominates the marginal harvesting productivity, the nuisance case, we obtain $a^* > a^\infty$. Hence, taking the nuisance into account yields less wildlife than not doing so, $X_0^* < X_0^\infty$.

The comparative static results are now found by taking the total differential of equations *(14)* and *(6)* (for details, see again Appendix 1). More nuisance increases the agricultural damage and, hence, reduces the marginal benefit from crop production. This direct effect is the same as in the no property rights case, and increases the equilibrium hunting effort. In addition there is now an indirect effect working through the stock size at the beginning of the next hunting season, which reinforces the direct effect. $\partial a^* / \partial\gamma$ is therefore stronger when it is local property rights. The direct effect of a higher game meat price is more benefit from hunting. As for the no property rights regime, this motivates for increased hunting and stock depletion, which is the same result as in the standard harvesting model (see, e.g., Clark 1973). Now, however, this effect is weakened because a higher offtake reduces the crop damage and hence, motivates for more effort allocated to crop production because the crop damage is taken into account. Thus, the indirect effect is of the opposite and the sign of $\partial a^{*} / \partial P_{H}$ is ambiguous. In the same way, the effect of a higher agricultural price is ambiguous. These results contrast the no property case, but are in line with results from harvesting models with competing uses among different assets (see, e.g., Swallow 1990). Other results are again given in Table 1, and as seen, the outcomes are to a large extent ambiguous.

6. Numerical illustrations

The theoretical reasoning will now be illustrated by data that fits reasonable well with the exploitation of the wildebeest population in the Serengeti-Mara ecosystem. This ecosystem covers an area of some 25,000 km^2 on the border of Tanzania and Kenya (Sinclair 1995). The Serengeti National Park is a part of it, and is wide known for the migration of its large herbivore populations of which the wildebeest has been most in the focus. Each year about a million wildebeest migrate across the Serengeti-Mara ecosystem (Murray 1995). The overall migratory pattern is supposed to be related to food supply, which in turn is connected to rainfall. The Serengeti ecosystem can be divided into two main regions; the southern short grasslands with low annual rainfall and the wooded northern grassland with higher rainfall (Fryxell 1995). The migratory wildebeest use the short grasslands in the south during the wet season and the tall grassland in the north during the dry season (Sinclair 1984 and 1995, Fryxell 1995).

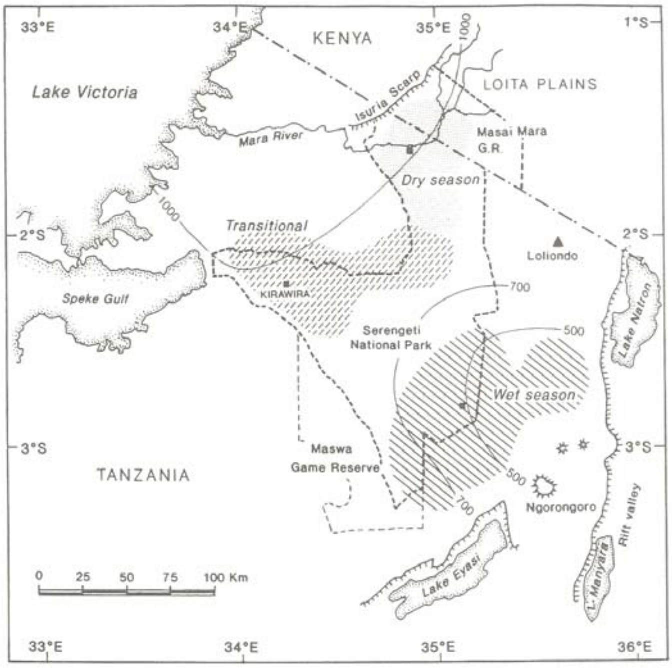


Figure 2: The seasonal ranges occupied by migratory wildebeest in the Serengeti-Mara ecosystem (adopted from Murray 1995).

The migratory herds know no boundaries, and make extensive use not only of the gazetted land, but also the open areas in the districts outside the borders of Serengeti National Park (cf. Figure 2). During the migration they spread beyond the park into the western frontier and enter land settled by humans. This side of Serengeti National Park is densely populated, and the population is increasing (Barrett and Arcese 1998). As a consequence, there are threats to the species diversity and the ecosystem because the landscape is modificated and habitat land is converted into agricultural use (Sinclair and Arcese 1995). The detrimental effects of the human-wildlife interaction are, however, not only one-way. The local people and the local communities also bear costs from the high wildlife abundance as the large herds of migrating herbivores induce crop damage (Emerton and Mfunda 1999). Villagers protect their crops and compensate themselves by hunting; in addition they also hunt for meat (Arcese et al. 1995). Hunting in this region is basically illegal[12]. Therefore, living with wildlife is twofold; it represents a

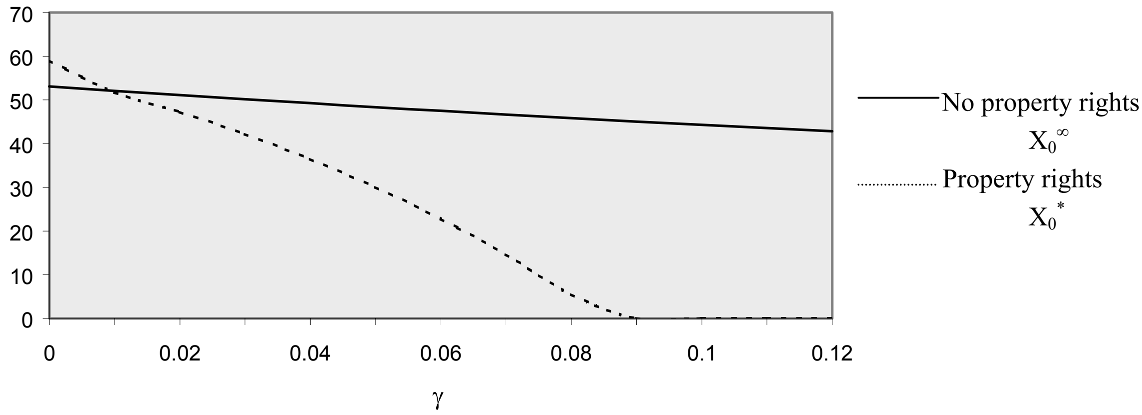
[12] Harvesting is not strictly illegal in parts of the western side of the Serengeti National Park, and persons having a car and firearms can obtain a hunting licence (Rugumayo 1999).

cost to the local peasants due to crop damage while it represents a benefit due to illegal harvesting.

The ecological model, using the Serengeti-Mara ecosystem as an illustration, is specified for the numerical analysis at the scale of one km^2 and one year. The same scale is also used for the agricultural benefit as well as the hunting benefit. The closer definition of the protected area and the outer area is found in Appendix 2. The baseline values for prices and costs together with ecological data and data for crop production used in the simulations are also presented here. As demonstrated above, the conservation effect of giving the local people property rights depends critically on the costs and benefits of living with wildlife. Because of the unclear nuisance effects, the damage coefficient γ will be varied throughout the simulations where a low value of γ , as mentioned, may be interpreted as more extensive use of fencing or other measures taken to protect crop production from the roaming wildlife. In addition, because of the unclear price effect under the property rights scenario, we will also vary the crop price P_A . The conservation effect of shifting the hunting price P_H together with changing ecological and productivity conditions will be studied as well.

Figure 3 demonstrates how the wildlife abundance varies with the crop damage coefficient γ under both property right schemes. More nuisance means less wildlife, and hence, the conservation-damage schedule slopes down under both scenarios. However, in accordance with the theoretical reasoning, the nuisance effect is quite more substantial under the local property rights regime. The figure also demonstrates that handing the property rights over to the local people gives more wildlife only when the crop damage is quite low. Hence, only when $\gamma < 0.01$ and less than 1% of the forage is eaten from the crop, local property rights results in higher wildlife abundance. Within this range the marginal benefit of living with wildlife exceeds the marginal cost, i.e., the term $\left[P_A\gamma\beta - P_H qa\right]$ from section five is negative. The figure also demonstrates that the nuisance effect is so strong that the presence of local property rights means a total depletion of the wildlife when more than 9 % of the forage is eaten from the crop. Hence, according to the model and the imposed parameter values where the baseline value of the damage coefficient is 12 %, $\gamma = 0.12$ (see Appendix 2), the presence of local property rights would have meant an extinction of the wildlife.

Figure3: The conservation-damage schedules.



Under the prevailing economic conditions, handing the property rights over to the local community is therefore not likely to promote wildebeest conservation in Serengeti. A better conservation strategy is probably to impose tighter damage control and, hence, reduce the cost of living with wildlife. The figure demonstrates, for instance, that a total eradication of crop damage from today's level (i.e., γ reduces from 0.12 to 0.00), increases wildlife abundance by somewhat above 20 % within the present property rights regime. This effect may further be strengthened if combined with, say, increased productivity and profitability in agricultural production as a higher crop price means more wildlife and conservation. This will be so within both regimes, and both conservation-damage schedules in Figure 3 shift up when P_A increases. However, contrary to what is expected from the theoretical reasoning, the effect is stronger under the property rights regime, and, for instance, we find that the same degree of conservation is met for $\gamma = 0.03$ when P_A is doubled from today's level. Shifting up the crop productivity works in the same manner while a higher game meat price P_H motivates for more hunting within both property rights schemes, and both conservation-damage schedules in Figure 3 shift down accordingly. A positive shift in the catchability coefficient q due to, say, improved hunting technology, also means reduced wildlife abundance within both property rights schemes, and the same degree of conservation is met for a higher value of the damage coefficient. We have also studied the effects of shifting the migration period *T*, interpreted as taken place as a result of, say, drought. The conservation-damage schedules, however, are only modestly influenced.

7. Discussion and concluding remarks

There is a growing recognition that a viable and sustainable wildlife utilization in the future depends on the support of the local communities living close to the wildlife, and during the last years community based wildlife management projects have increasingly become one of the means for safeguarding wildlife in sub-Saharan Africa (Kiss, 1990, Swanson and Barbier 1992). The basic idea behind these schemes is to engender the co-operation of local communities in wildlife conservation and wildlife management by ensuring that parts of the benefits from wildlife utilization go to the local communities. There are several ways in which benefit sharing can take place; through revenue sharing from tourism, safari hunting, or establishing user rights through hunting quotas (see Skonhoft 1998 for an analysis), or through local job creation in tourism, wildlife and park services. The experiences from two prototypes of community based projects are summarised by Kiss (1990), namely Windfall and Campfire (both in Zimbabwe).

What has been analysed in this paper is a more far-reaching community based management system as the local people have been given the full property rights of the wildlife; that is, they control and obtain the *whole* benefit stream from the wildlife when it is outside the protected area. This management scheme has been compared to the polar one of having no property rights. Under the scenario of no property rights, the local people have no incentive to take into account that their hunting today influences the wildlife stock and hence, reduces the potential for hunting next year. Technically, the stock size is then treated as an exogenous variable when allocating hunting effort, and is in line with the 'open-access' solution of Smith (1975). In the next step, when assuming that the property rights are handed over to the local people, they have incentives to invest in the wildlife stock, and take the stock size into account when hunting. Smith (1975) specifies this scenario within an inter-temporal framework where the present-value benefit is maximized. The present exposition where the local people consider the wildlife stock as an endogenous variable under ecological equilibrium coincides with the steady-state of the Smith model when the discount rate is zero (see also Munro and Scott 1985). Throughout the analysis it is assumed that the local people is a homogenous group with no internal conflicting interests. That is, the property rights scenario is implemented in its most optimistically way.

The main point of the analysis is to find under what conditions community based management and property rights of the local people might result in a higher wildlife abundance compared to the no property rights scenario. It is shown that the actual outcome depends critically on the difference between the benefit and cost of living with wildlife; that is, the marginal harvesting value versus the marginal crop damage. Accordingly, only when the nuisance from the roaming wildlife is small and the marginal benefit exceeds the cost, we find that community based management increases the wildlife stock compared to the scenario of no property rights. The main message from the theoretical analysis is therefore that relying on local property rights alone as a tool in wildlife conservation may not work. This conclusion contrasts Kiss (1990) and Swanson and Barbier (1992) who argue that local property rights generally will promote wildlife conservation.

The theoretical model is illustrated by data that represents the Serengeti-Mara ecosystem in a stylised manner. For the baseline parameter values the nuisance from the wildlife dominates the benefit, and hence, local property rights may give incentives to deplete the roaming stock of wildebeest. A better conservation strategy than handling the property rights over to the local people is probably to protect crop production by supporting fencing or take other types of control measures to reduce the damage caused by the roaming wildlife. The numerical simulations also demonstrate that an increased crop price or higher crop productivity through, say, improved fertiliser and pesticides use, may lead the local people to channel more effort towards agricultural production. Hence, such measures will also work in the direction of more wildlife conservation. This conclusion is in line with Barrett and Arcese (1998) who find that to succeed promoting wildebeest conservation in Serengeti, wildlife management schemes must attempt to increase local people's benefit from alternative activities.

The focus of the present analysis has been property rights and wildlife conservation. The basic idea behind giving the local people full property rights over the wildlife is, however, to promote wildlife conservation *together* with economic development and increased welfare (see, e.g., Kiss 1990). *Ceteris paribus*, the presence of property rights increases the welfare of the local people compared to the scenario of no property rights as both effort and wildlife abundance are adjusted optimally within this regime, while only effort is adjusted within the no property rights regime. When the damage of the roaming wildlife is small so that the benefit of living with wildlife exceeds the cost, we can therefore conclude that the presence of local property rights promotes wildlife conservation as well as local welfare. On the contrary, according to our numerical results, local welfare is promoted at the expense of wildlife conservation when the cost of living with wildlife exceeds the benefit.

References

Arcese, P., J. Hando, and K. Campbell (1995): "Historical and present-day poaching efforts in Serengeti". In Sinclair, A.R.E. and P. Arcese (eds): *Serengeti II*, University of Chicago Press, Chicago, 506-533.

Barrett, C.B. and P. Arcese (1998): "Wildlife harvest in integrated conservation and development projects", *Land Economics*, 74(4), 449-465.

Bromley, D. (1991): *Environment and Economy*. Blackwell, Oxford.

Brown, K. et al. (1993): "Economics and the conservation of global biological diversity", *Global Environmental Facility, Working Paper*, No. 2, The World Bank, New York.

Bulte, E. and G. van Kooten (1999): "The ivory trade ban and elephant numbers: Theory and application to Zambia", *American Journal of Agricultural Economics*, 81(4), 453-466.

Campbell, K. and H. Hofer (1995): "People and wildlife: Spatial dynamics and zones of interaction". In Sinclair, A.R.E. and P. Arcese: *Serengeti II*, University of Chicago Press, Chicago, 534-570.

Clark, C.W. (1990): *Mathematical Bioeconomics*, second edition, Wiley, New York.

Clark, C.W. (1973): "Profit maximization and the extinction of animal species", *Journal of Political Economy*, 81(4), 950-961.

Caughley, G. and A. Sinclair(1994): *Wildlife Ecology and Management,* Oxford University Press, Oxford.

Emerton, L. and I. Mfunda (1999): "Making wildlife economically viable for communities living around the western Serengeti, Tanzania", *Evaluating Eden Series Discussion Paper*, No. 1, International Institute for Environment and Development, London.

Fancy, S., K. Whitten and D. Russel (1994): "Demography of the porcupine caribou herd", *Canadian Journal of Zoology*, 72(5), 840-846.

Fryxell, J.M. (1995): "Aggregation an migration by grazing ungulates in relation to resources and predators". In Sinclair, A.R.E. and P. Arcese (eds): *Serengeti II*, University of Chicago Press, Chicago, 257-273.

Getz, W. and R. Haight (1989): *Population Harvesting*, Princeton University Press, Princeton.

Gordon, H.S. (1954): "The economic theory of a common property resource: The fishery", *The Journal of Political Economy*, 62, 124-142.

Homans, F.R. and J.E. Wilen (1997): "A model of regulated open access resource use", *Journal of Environmental Economics and Management*, 32(1), 1-21.

Kiss, A. (1990): "Living with wildlife: Wildlife resource management with local participation in Africa", *Technical Paper*, 130, World Bank, Washington D.C.

Mangel, M. et al. (1996): "Principles for the conservation of wild living resources", *Ecological Application*, 6(2), 338-362.

Marks, S.A. (1984): *The Imperial Lion: Human Dimensions of Wildlife Management in Africa*, Westview Press, Boulder.

Munro, G. and A. Scott (1985): "The economics of fishery management". In Kneese, A. and J. Sweeney (eds): *Handbook of Natural Resource and Energy Economics*, 2, Elsevier Science, Amsterdam, 623-673.

Murray, M.G. (1995): "Specific nutrient requirements of wildebeest". In Sinclair, A.R.E. and P. Arcese (eds): *Serengeti II*, University of Chicago Press, Chicago, 231- 256.

Naughton-Treves, L., and S. Sanderson (1995): "Property, politics and wildlife conservation", *World Development*, 23(8), 1265-1275.

Rugumayo, C.R. (1999): "The participation in sustainable resource management. The case of Serengeti regional conservation strategy, Tanzania", Working Paper 112, Norwegian Institute for Urban and Regional Research, Oslo.

Sinclair, A.R.E. (1984): "The function of distance movements in vertebrates". In Swingland, I.R. and P.J. Greenwood (eds): *The ecology of animal movement*, Clarendon Press, Oxford.

Sinclair, A.R.E. (1995): "Serengeti past and present". In Sinclair, A.R.E. and P. Arcese (eds): *Serengeti II*, University of Chicago Press, Chicago, 3-30.

Sinclair, A.R.E. and P. Arcese (1995): "Serengeti in the context of worldwide conservation efforts". In Sinclair, A.R.E. and P. Arcese (eds): *Serengeti II*, University of Chicago Press, Chicago, 31-46.

Skonhoft, A. (1998): "Resource utilisation, property rights and welfare. Wildlife and the local people", *Ecological Economics*, 26(1), 67-80.

Skonhoft, A. and J.T. Solstad (1998): "The political economy of wildlife exploitation", *Land Economics*, 74(1), 16-31.

Smith, V.L. (1975): "The primitive hunter culture, pleistocene extinction, and the rise of agriculture", *Journal of Political Economy*, 83(4), 727-755.

SRCP (1998): "Overview of the communities surrounding the Ikorongo/Grumeti Game Reserves", Report for the Ministry of Natural Resources and Tourism, Arusha, Tanzania.

Swallow, S. (1990): "Depletion of the environmental basis for renewable resources. The economics of interdependent renewable and non-renewable resources", *Journal of Environmental Economics and Management*, 19(3), 281-296.

Swanson, T.M. and E.B. Barbier (1992): *Economics for the Wilds: Wildlife, Wildlands, Diversity and Development*, Earthscan, London.

TANAPA (1996): "Serengeti national park. Management zone plan/environmental impact assessment", Tanzanian National Parks, Tanzania.

Wright, R. (1999): "Wildlife management in the national parks: questions is search of answers", *Ecological Applications*, 9, 30-36.

Zivin, J., B.M. Hueth and D. Zilberman (1998): "Managing a multiple-use resource: The case of feral pig management in California rangeland", *Journal of Environmental Economics and Management*, 39(2), 189-204.

Appendix 1

Comparative static results

The comparative static of the no property rights scenario of the local people is found by taking the total differential of *(6)* and *(13)*. The result is

$$\begin{bmatrix} 1 & -[1+r-2/s]KqT/rs \\ P_H qTs + P_A\gamma\beta C/a & Q \end{bmatrix}\begin{bmatrix} dX_0 \\ da \end{bmatrix} = \begin{bmatrix} 0 \\ -P_A\beta X_0 C/a \end{bmatrix}d\gamma$$

(A1)

$$+\begin{bmatrix} 0 \\ -qX_0Ts \end{bmatrix}dP_H + \begin{bmatrix} 0 \\ A'(N)T-\gamma\beta X_0 C/a \end{bmatrix}dP_A + \begin{bmatrix} qaK[1+r-2/s]/rs \\ G \end{bmatrix}dT$$

where $Q=-P_H X_0(qT)^2 s + P_A A''T^2 + P_A\gamma\beta X_0 qTs/a < 0$ from the second order maximum conditions while the sign of $G = P_A A'(N) - P_A A''(N)aT - s(1-qaT)X_0[P_H qa - P_A\gamma\beta]/a$ is ambiguous. The determinant of the system $Q + KqT[P_H qTs + P_A\gamma\beta C/a][1+r-2/s]/rs$ is also negative due to the second order conditions.

The comparative static results when having property rights are found when taking the total differentiation of *(14)* and (*6)*. Equations *(A2)*-*(A4)* give the stock effects where $Z > 0$ from the second order maximum condition.

$$(A2)\quad dX_0/d\gamma = P_A\beta[X_0C-(1-s)TK[(r+1)-2/s]/rs]qTK[(r+1)-2/s]/Zras < 0$$

$$(A3)\quad dX_0/dP_H = qT[X_0s+(1-s)K[(r+1)-2/s]/rs]qTK[(r+1)-2/s]/Zras$$

$$(A4)\quad dX_0/dP_A = \begin{Bmatrix} \gamma\beta X_0C - \gamma\beta T(1-s)TK[(r+1)-2/s]/rs \\ -A'(N)aT \end{Bmatrix} qTK[(r+1)-2/s]/Zras$$

Appendix 2

The numerical analysis

As mentioned in the main text, the ecological model is specified for the numerical analysis at the scale of one km^2 and one year. The same scale is also used for the agricultural benefit, as well as the hunting benefit, given in 1998/99 prices. The protected area, consisting of Serengeti National Park and its surrounding game reserves, covers an area of some 26 000 km^2 (TANAPA Planning Unit 1996). The outer area is considered the catchment area surrounding the western edge of the protected area. Following Campbell and Hofer (1995), we assume that the catchment area, where the local people are originated, is located with a maximum distance of 45 km to the protected area. This area constitutes some 30 500 km^2 (Campbell and Hofer 1995). The human population in the outer area is estimated to be about 1.1 million with an average household size of about 7 persons (Campbell and Hofer 1995). Accordingly, there will be about 5 households per km^2 in the outer area. On average, 2 persons per household work in agricultural production and hunting. Hence, the effort constraint L yields 10•365 days a year, and we have $3650 = N + aT$.

The wildebeest migration is related to rainfall and the wildebeest population enters agricultural land while it grazes the northern grassland during the dry season. The length of the dry season is about four months (Barrett and Arcese 1998). Four months, or 122 days, is also assumed to be the length of the migration period, and hence, T=122 days. The wildebeest population is estimated to be about 1.3 million animals and the annual offtake to some 120 000 animals (Campbell and Hofer 1995). The wildlife density during the period of migration is therefore about 43 animals per km^2, while the total offtake is 4 animals per km^2. The number of hunters per household in the western Serengeti is assumed to be about 0.2 (Campbell and Hofer 1995). Consequently, it is one person involved in hunting at full time basis for every 5 households, and the baseline value of a is accordingly 1. By imposing these values for a and T into the survival rate together with H / X_0 as 4/43, the catchability coefficient is calculated from equation *(4)*. The result is q = 0.0008. Following Barrett and Arcese (1998), the hunting value of a wildebeest is about US\$16, $P_H = 16$. The natural growth function is specified as logistic. To calibrate the wildlife stock to its base level the carrying capacity K is set to 59 animals per km^2 outer area, meaning that the ecosystem can carry a stock of wildebeest just below 1.8 million animals. The maximum specific growth rate is fixed as $r = 0.3$ (Caughley and Sinclar 1994).

The yield function is specified as $A(N) = \mu N^{\alpha}$ with μ as a productivity parameter and α as a scale parameter. The scale parameter is given as 0.8 (Barrett and Arcese 1998). According to a questionnaire among 300 households in Serengeti and Bunda Districts in June-September 2001 (Anne Borge Johannesen, work in progress), the average plot size per household is 7.4 acres, corresponding to a cultivated area of a fraction of 0.15 per km^2 for the average 5 households. For the same districts, the average value of the yield is estimated to be about US$19 000 per km^2 cultivated land (Emerton and Mfunda 1999). At our scale of one km^2, this represents a value of 19.000•0.15 = US$2 850 (or US$570 per household). The main crops grown in the Western Serengeti are sorghum, cassava, maize, and cotton (SRCP 1998). Personal communication (SRCP 1999) indicates a per kg price of US$ 0.18 for sorghum, US$ 0.05 for cassava, US$ 0.11 for maize, and US$ 0.19 for cotton. By weighting the crop prices by the relative magnitude of these crops, we arrive at the price per kilo agricultural output equals US$ 0.15. P_A = 0.15. Consequently, for the specified yield function we have $P_A \mu N^{\alpha} = 0.15 \bullet \mu(3650 - 1 \bullet 122)^{0.8} = 2\,850$, balancing with $\mu = 28$.

The wildebeest has a daily consumption of about 3% of their body weight in dry forage (Murray 1995), hence, $\beta = 0.03$. When assuming an average weight of 150 kg per animal, the daily consumption of dry forage is therefore 4.5 kg per animal. To obtain a value for the fraction of the forage eaten from the crops γ, we use the first order condition *(13)*. For the above estimated parameter values together with the imposed values for a and X_0, $\gamma = 0.12$ balances this equation. Hence, through this calibrating we find that 12% of the wildebeests daily consumption of dry forage is from crops. Plugging into equation *(8)* this corresponds to a yearly damage on crop production of about 14%. This is the average of the estimates found in Emerton and Mfunda (1999) and Anne Borge Johannesen (work in progress). Table A1 summarises the baseline parameter values.

Table A1: Baseline values prices and costs (1998/99-prices), ecological parameters and other parameters

Parameter	Description	Value
P_H	Meat price	16 ($/animal)
q	Catchability coefficient	0.0008 (1/day)
P_A	Crop price	0.15 ($/kg)
α	Input elasticity labour crop production	0.8
μ	Productivity crop production	28 (kg/day)
β	Fraction daily consumption of dry forage	0.03
γ	Fraction dry forage consumption crop	0.12
L	Total available effort	3650 (day)
T	Length of migration period	122 (day)
K	Carrying capacity	59 (animal)
r	Intrinsic growth rate	0.3

Chapter 4

Designing Integrated Conservation and Development Projects: Hunting incentives and human welfare with numerical illustrations from Serengeti

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Abstract:

Integrated Conservation and Development Projects (ICDPs) attempt to link conservation of natural resources in protected areas to economic development in the surrounding communities. Such projects have been introduced in many parts of Africa, but their performance has so far been hampered by numerous difficulties. This paper develops a hunter-agrarian household model to explore the effect on wildlife conservation and human welfare of the most common instruments of existing ICDPs. It is demonstrated that stimulating working opportunities in the formal sector has the potential of promoting conservation and welfare, while money transfers and distribution of game from managed culling fail in conserving wildlife, if not explicitly linked to the conservation objective. In contrast, the analysis shows that such links, modelled as a risk of being excluded from the project components if caught in illegal hunting, may be a more durable mean for ICDPs to reach its goal of improved wildlife conservation and human welfare. The model is illustrated by numerical calculations with data from Serengeti.

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1. Introduction

1.1. Background

Integrated Conservation and Development Projects (ICDPs) are widely launched as the solution to the problem of biodiversity loss in developing countries. In sub-Saharan Africa, ICDPs are frequently designed to encourage conservation by reconciling the management of protected areas with the social and economic needs of the local people (see Kiss 1990, Barbier 1992, Brandon and Wells 1992, Wells and Brandon 1992, Barrett and Arcese 1995, Barrett and Arcese 1998, Songorwa 1999). Through benefit sharing it is expected that ICDPs will discourage poaching and promote economic development (see Kiss 1990 and Barbier 1992). The understanding is therefore that such a management scheme can improve the livelihood of rural communities without contributing to environmental degradation. There are several ways in which ICDPs can generate benefits for the local people, e.g. through revenue transfers from tourism, local job creation in the formal sector, stimulating increased productivity in the agricultural sector, and so forth. Benefit sharing is also obtainable through direct utilization of wildlife, such as harvesting quotas for the local communities and controlled culling operations. By providing such benefits, the ICDPs aspire to stimulate the local people to reduce wildlife exploitation (Brandon and Wells 1992, Wells and Brandon 1992, Gibson and Marks 1995, Songorwa 1999).

However, ICDPs have recently attracted attention because of the untested assumptions behind the projects. Brandon and Wells (1992) give a broad and instructive discussion of the design dilemmas of ICDPs and describe some of the trade-offs inherent in linking conservation and development. Experience shows that many existing projects lack a direct link between the hunting activity of the local people and the conservation objective. As pointed out by Kiss (1990) and Brandon and Wells (1992), without such a link it is difficult for the local people to realize that there is a purpose of improved conservation behind the benefits they receive. If worst comes to worst, they may regard the benefits as lump-sum transfers and carry on the exploitation activities as before. Both Kiss (1990) and Brandon and Wells (1992) stress the necessity of establishing such a connection.

The lack of a proper link in existing ICDPs is the point of departure for the present paper. The main purpose is to compare the performance of two different ICDP designs or regimes. The difference between the regimes is related to the implementation of one of the four ICDP tools of focus here. This tool is the benefit-sharing instrument which consists of distribution of game meat from controlled culling and income transfers from the tourism sector. In the first regime, an ICDP is implemented without an explicit link between the benefit transfers and the hunting activity of the local people. That is, there is no risk of being excluded from these benefits if caught in illegal hunting. This corresponds to Barrett and Arcese's model (1998) (see below) and is in line with the design of most existing ICDPs today (see Brandon and Wells 1992). In the second regime, the project designer implements a link by creating a continuous risk of being excluded from the benefit transfers if caught in illegal hunting. Then, if caught, the local people receive no benefits from the ICDP. There are, of course, other ways in which ICDPs can link the benefit transfers with the conservation objective. For instance, ICDPs may offer comprehensive training and education to the local people in order to make them realize that the magnitude of future transfers depends on the wildlife abundance – and consequently on their hunting activity – as more wildlife generates more income in tourism and opens for more extensive game culling. The link modelled here, however, is more direct and easier to implement because the performance of the project does not require that the local people behave in a self-enforcing and less myopic way.

Barrett and Arcese (1998) reveal possible unintended outcomes of benefit sharing for wildlife conservation. They present a bio-economic model with no explicit link between the conservation objective and the benefit-sharing instruments. The economic part of the model consists of a representative household for the rural population in western Serengeti in Tanzania which derives utility from consumption of game meat, agricultural output and leisure. It is also assumed that no market exists for game meat, meaning that the meat is used for household consumption. First, they demonstrate that money transfers from tourism may lead to a smaller degree of wildlife conservation. This is because the money transfers produce a positive income effect on game meat consumption. Second, they show that the conservation effect of game meat distribution from controlled culling is negative. Because the market for game meat fails, increased endowment of meat leads the household to reduce the illegal hunting. However, game meat consumption is considered a normal good and, consequently, there is a less than one-for-one trade-off of distributed meat for illegal meat, meaning that the aggregate wildlife offtake increases[1]. Hence, the analysis of Barrett and Arcese (1998) gives reasons to question ICDPs that rely on money transfers and culling operations[2].

As already mentioned, the main focus of this paper is to analyse under what conditions, and for which design, ICDPs relying on benefit-sharing instruments can promote wildlife conservation and human welfare. Quite similar to Barrett and Arcese (1998), the analysis demonstrates that, in absence of a link between illegal hunting and the received transfers, as described above, benefit transfers will lead to a smaller degree of wildlife conservation. However, in contrast to Barrett and Arcese (1998), benefit sharing may promote wildlife conservation in presence of a link between the transfers and illegal hunting. It turns out that benefit transfers perform better if combined with more extensive use of guards and patrol units in the protected area. In this case, benefit sharing may also improve the economic conditions of the local community.

In addition to the design of the benefit-sharing strategies, the present paper also looks at the role of working opportunities in the formal sector, improved productivity in agriculture and, as mentioned, anti-poaching law enforcement. The wage rate in the formal sector and the productivity in agriculture are considered independent of whether the local people are caught in illegal hunting. This assumption seems reasonable because ICDPs cannot fully control the conditions in these sectors. In the analysis of anti-poaching law enforcement the focus is on increased use of guards and patrols, in

[1] This result follows automatically from the assumption that game meat is non-tradable, i.e. the consumption C equals the sum of illegal meat h and distributed meat M as $C = h + M$. Partial derivation of this constraint with respect to M gives $\partial C / \partial M = \partial h / \partial M + 1$. Because game meat is a normal good, i.e. $\partial C / \partial M > 0$, it follows from this equality that $\partial h / \partial M > -1$.

[2] When it comes to game meat distribution, the opposite result occurs in Skonhoft's model (1998). This stems from the fact that Skonhoft considers the local people as passive, while the park manager, who benefits from both legal hunting and tourism, is the active agent. When the state instructs the park manager to distribute a fraction of the legal harvest to the local people, the marginal return on hunting is reduced relative to the marginal return on tourism and, consequently, the manager increases the investment in wildlife. With a passive local community, this promotes wildlife conservation.

order to improve the detection rate, and an increase in the fine level imposed on the local people if they are caught.

When it comes to agricultural policies, attempts to stimulate productivity improvements are repeatedly suggested in wildlife management (see Brown et al. 1993). It is believed that this will divert labour away from wildlife hunting to agricultural production. However, Lopez (1998) demonstrates that this argument may be false, depending on for what type of crop the productivity increases; i.e. land-intensive crop or labour-intensive crop. Using a static model with fixed land endowments and no labour market, he shows that an increased price for the land-intensive output is likely to reduce the labour demand for farming, and hence increase the resource extraction. Also Schulz and Skonhoft (1996) discuss agricultural productivity and its impact on resource extraction. Focusing on the conflict between land as an input agricultural production and land for wildlife habitat, they demonstrate that a higher return on agriculture increases the conversion of land and is therefore a threat to wildlife conservation (see also Skonhoft 1999).

Other contributors who question the importance of agricultural productivity are Bulte and van Soest (1999). Utilizing a dynamic model for a hunter-agrarian household, they demonstrate that the conservation effect of increased agricultural productivity is unclear and critically dependent on whether or not there exist markets for game meat and labour. With such markets present, the household solves the optimal effort in agriculture and hunting separately[3]. Consequently, there is no effect on wildlife exploitation of improved productivity in agriculture. However, with no markets present, they demonstrate that the conservation effect of productivity improvements is ambiguous. These analyses indicate that ICDPs promoting agricultural productivity should be implemented with care, especially when it comes to land use conflicts and regional conditions such as the functioning of markets.

[3] The intuition is that when a labour market is present the household is able to alter the effort use in agriculture by adjusting its labour supply in formal employment, while the effort use in wildlife harvesting is left unchanged, and vice versa.

The following analysis differs from the contributions quoted above in two important ways. First, while most of the authors referred to above limit their focus to benefit-sharing strategies and improved agricultural productivity, this paper also considers the impact of employment in formal sector and anti-poaching law enforcement. The main contribution of the paper is to suggest an ICDP design that will succeed in promoting wildlife conservation. As already mentioned, this involves an explicit link where the local people are excluded from the benefit transfers if caught in illegal hunting. Second, because ICDPs also aim at improving the economic conditions of the local people, the present paper extends the above contributions by exploring the welfare effect of the ICDP incentives.

1.2. Assumptions

This paper presents a bio-economic model of a hunter-agrarian community on the border of a protected area, i.e. a national park. The property rights to wildlife are held by the State which has appointed a park agency to manage the protected area. The local people have no legal rights to wildlife hunting, but the degree of anti-poaching law enforcement imposed by the park agency is not sufficient to eliminate illegal hunting and effectively protect the property rights of the State[4].

Throughout the analysis, the local people are considered the only active agent, while the park authority is passive (see also Milner-Gulland and Leader-Williams 1992). This means that the State instructs the benefit-sharing strategies and the law enforcement activities on the park manager and, hence, these activities are implemented as exogenous[5]. This is contrary to the analyses of Skonhoft (1995), Skonhoft (1998), and Skonhoft and Solstad (1998) where the game culling and the level of law enforcement are determined within the model. Following Barrett and Arcese (1998), it is further assumed that game meat is distributed freely to the local communities and, consequently, this activity does not generate income for the park manager. However, the park agency benefits from non-consumptive use of the wildlife, such as tourism. A fixed share of this income – pre-determined by the State – is transferred to the local people.

[4] For a broad discussion of protected areas and law enforcement, see Martin (1993).
[5] See e.g. Wright (1999) for a further discussion of culling operations as a tool in wildlife management.

As in Barrett and Arcese (1998), Lopez (1998), and Bulte and van Soest (1999), it is assumed that illegal hunting is only performed by the local people living in the vicinity of the protected area. Poaching carried out by outsiders is therefore ignored. In addition to illegal hunting, the local people are involved in agricultural production and offered work opportunities in the formal sector, such as employment in the local industry and tourism. A market for labour is therefore present and this is in accordance with the general model of Bulte and van Soest (1999), while Barrett and Arcese (1998) and Lopez (1998) assume that no such market exists. In contrast, there exists no market for game meat which may be a result of high transaction costs (see e.g. Sadoulet and de Janvry 1995). With no market for game meat the local people's consumption is constrained by the transfers from the managed culls and the illegal offtake (see below, see also Barrett and Arcese (1998)).

The rest of the paper is organized as follows. The model is presented in section 2, while section 3 explores the conservation and welfare effects of ICDPs implemented without a link between the benefit sharing instruments and illegal hunting. That is, the local people receive the benefit transfers regardless of their involvement in illegal hunting. This regime is referred to as Regime I. In Regime II in section 4, the model is adjusted to investigate the performance of ICDPs where the local people face a risk of being excluded from the benefit transfers if caught in illegal hunting. Section 5 illustrates the theoretical model with a numerical example of the wildlife exploitation in Serengeti. The paper is closed by a discussion and concluding remarks in section 6.

2. The model

Consider a local community consisting of a homogeneous group of peasants living on the border of a protected area[6]. The local people produce two types of output; agricultural crops and game meat. The agricultural production is dependent on the amount of agricultural land, pesticides and fertiliser use, rainfall etc., as well as labour

[6] There are assumed to be conflicting interests among the local people. Hence, prevalence of individual conformity to group norms is assumed to be present. In line with traditional reasoning, it is assumed that the elders are in charge of the group's activities (Marks 1984).

effort use. Following Barrett and Arcese (1998), Lopez (1998) and Bulte and van Soest (1999), all factors are fixed except for labour E_A. Then, the production function reads

$$(1) \qquad A = A(E_A),$$

where output increases by a decreasing rate in effort, $A' > 0$ and $A'' \leq 0$.

The households produce game meat through illegal hunting of wildlife in the protected area. It is assumed that the wildlife offtake is a function of labour effort and the wildlife abundance, specified as

$$(2) \qquad h = f(E_h)X,$$

where E_h is effort directed towards hunting and X is the wildlife stock. The offtake increases by a decreasing rate in effort, $f' > 0$ and $f'' \leq 0$[7]. The degree of effort directed towards wildlife hunting is influenced by the fact that this activity is illegal. The probability of being caught in illegal hunting θ is given as

$$(3) \qquad \theta = min\,[1, \delta E_h]$$

θ increases with the hunting effort, but the probability of being caught cannot exceed 1 (see also Skonhoft and Solstad 1998). $\delta > 0$ is the marginal rate of detection, which reflects the productivity or the level of law enforcement activities carried out by the park manager. For a given hunting effort, more extensive enforcement use increases δ and, hence, θ increases. If caught in illegal hunting, the households are imposed a fixed fine F.

In addition to wildlife hunting and agricultural production, the local people have the opportunity to work in the formal sector (see section 1.2). Let N be the labour use in

[7] Concavity of f may be due to technological restrictions such as access to weapons, transport etc. It is seen that if $f'' = 0$, *(2)* is in line with the Schaefer harvesting function.

formal sector and T be the fixed available labour effort interpreted as the total human population living in the vicinity of the conservation area[8]. Then, the time constraint reads

$$(4) \qquad E_h + E_A + N \leq T$$

Throughout the analysis the time constraint is assumed to be binding, and hence, there is always a positive opportunity cost of labour use. As a result, a trade-off between effort in wildlife hunting and the legal activities is present.

The households derive utility from the consumption of agricultural output and game meat. There are two possible states in this model: either the local people will manage to escape the anti-poaching patrols with the probability $(1-\theta)$, or they will be caught with the probability θ. It is assumed that the decision problem of the local people is to maximize the expected utility given as

$$(5) \qquad EW = (1-\theta)\left[U(C_A^e) + V(C_G^e)\right] + \theta\left[U(C_A^c) + V(C_G^c)\right]$$

Here, superscript '*e*' denotes the resulting consumption levels if the local people manage to escape the patrol units, while superscript '*c*' denotes the consumption levels if caught in illegal hunting. In order to simplify the analysis, the expected utility is specified as separable in agricultural and game meat consumption, i.e. C_A and C_G respectively, where $U'(\cdot) > 0$ and $V'(\cdot) > 0$. The magnitude of the second order derivatives of U and V reflects the local people's attitude towards risk. In general, the attitude towards risk depends on how wealthy the decision-maker is (see e.g. Dasgupta 1993, chapter 8). It is plausible that poor households would be more averse to accepting additional risk compared to relatively more wealthy agents, because the disadvantage of

[8] N is endogenous in this model. One may argue that there are constraints on the working hours in formal employment, which makes it difficult for individuals to adjust the working hours to changes in the wage rate in that sector. However, T is interpreted as the size of the human population and, hence, changes in N are due to individuals choosing not to work or to enter employment in the formal sector without altering the individual working hours.

the risk, here represented by the monetary punishment, is particularly harsh for households who have little to fall back on. Because this analysis is related to poor communities in remote areas in developing countries, it is assumed that the local people are risk averse[9]. This means that the expected utility function is strictly concave in agricultural and game meat consumption, i.e. $U''(\cdot) < 0$ and $V''(\cdot) < 0$.

A trade-off in hunting effort is present in the expected utility function *(5)*. First, more hunting effort increases the consumption of game meat which, in turn, increases the utility for a given consumption of agricultural output and probability of being caught. However, more hunting effort reduces the income from formal employment which restricts the agricultural consumption. This works in the direction of reduced utility. In addition, the realized consumption levels are, in general, higher if the local people manage to escape the patrol units, i.e. $C_A^e \geq C_A^c$ and $C_G^e \geq C_G^c$ (see below). Therefore, as more hunting effort increases the probability of being caught it reduces the expected utility level. The local people must consider these trade-offs when determining the allocation of labour effort between agricultural production, wildlife hunting, and employment in the formal sector.

The next step is to specify the prevailing constraints on consumption if the local people manage to escape the patrol units. First, the consumption of game meat consists of the illegal offtake and legal game meat distributed from the managed culls. In the following, these are considered homogenous. The extent of the game meat distribution is set as a fraction $m > 0$ of the wildlife stock X[10]. It is assumed that the local community in consideration is the only community to receive game meat from the culling program, which means that the whole amount *mX* is transferred to this

[9] We assume that the consumption level does not fall below a particular threshold or subsistence level at which point the local people would be attracted to the risk in order to avoid disaster.
[10] In the case of Serengeti the quota (*mX*) is set low relative to the wildlife stock for each village receiving meat from the culling program, meaning that *m* is low in the culling of today (see Appendix 2).

community[11]. Because there is no market for game meat, the consumption of meat if not caught in illegal hunting is constrained as in *(6)*[12].

$$(6) \qquad C_G^e = f(E_h)X + mX$$

The consumption of agricultural output if not caught in illegal hunting is constrained by the level of agricultural production, income from formal employment, and the income transfers from tourism. The latter is set as a fixed fraction $\mu \in [0, 1]$ of the net income in the tourism sector *S(X)*. It is assumed that *S(X)* increases with the wildlife density as the number of tourists increases, but at a decreasing rate $S' > 0$, $S'' \leq 0$ (see Bulte and van Kooten 1996). Employment in formal activities is paid by the exogenous wage ω. Let P_A be the unit price of agricultural output. Then, if the local people manage to escape the patrol units, the consumption expenditure on agricultural output equals $P_A C_A^e = P_A A(E_A) + \mu S(X) + \omega N$. Solving this equation with respect to C_A^e yields

$$(7) \qquad C_A^e = A(E_A) + \mu S(X)/P_A + \omega N/P_A$$

In general $C_A^e > A(E_A)$ which means that the local people will buy excess agricultural food on the market if they escape the patrol units. However, in absence of formal employment and an ICDP program (i.e. $N = \mu = 0$) the constraint in *(7)* reads $C_A^e = A(E_A)$. In this case, the agricultural consumption is constrained by the production level in this community.

The prevailing constraints on agricultural and game meat consumption if caught in illegal hunting are strictly dependent on the design of the ICDP. Sections 3 and 4 below outline in detail the resulting constraints for the respective regimes. The final step in this section is to present the population dynamics of wildlife. As already noted, the local

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[11] For a discussion of a broader distribution scheme, see Brandon and Wells (1992).

[12] The absence of a market for game meat captures the nature of the village economy in this commodity in Serengeti (see also Barrett and Arcese 1998). While there is trade in meat among households within and between villages in the catchment area, there is a small and negligible trade outside the catchment area.

peasants are the only agents involved in illegal hunting, meaning that their hunting constitutes the total illegal offtake. The natural growth of the population is specified as logistic, while the stock shrinks according to illegal hunting and managed culling, as given in equation *(8)*.

$$(8) \qquad dX/dt = rX(1 - X/K) - f(E_h)X - mX$$

Here, r is the intrinsic growth rate and K is the carrying capacity of the protected area. The total harvest equals the sum of the illegal offtake $f(E_h)X$, and the managed harvest mX. The ecological equilibrium is defined by a constant wildlife stock over time. Solving for X at $dX/dt = 0$ yields $X = (r - f(E_h) - m)K/r$, so that wildlife abundance is reduced by m and E_h. To obtain $X \geq 0$, the man made mortality must be restricted by $f(E_h) + m \leq r$ [13].

Before we turn to the specific ICDP regimes, we need to establish how the local people adapt to the ecology. As already noted, the State has the property rights to wildlife, while hunting performed by the local people is illegal. The local people experience, through the property rights scheme and anti-poaching law enforcement, a continuing risk of being effectively denied access to hunting. It is therefore assumed that their behaviour is based on short-term expected utility maximization and hence, they do not take the stock density into account when maximizing expected utility. Technically, this means that the local people treat the stock density as an exogenous variable and this corresponds to one of Smith's models (1975). See also Lopez (1998), Barrett and Arcese (1998), and Skonhoft and Solstad (1998).

3. Regime I

In this regime, the ICDP manager transfers money from tourism and game meat to the local people independent of whether they are caught in illegal hunting. Hence, if caught in illegal hunting, the resulting budget available for agricultural consumption is lowered

[13] Hence, the ecological equilibrium restricts the size of m: as X approaches zero $m = r - f(E_h)$.

by the imposed fine F only. The constraint on agricultural consumption is therefore given by

$$(9) \qquad C_A^c = A(E_A) + \mu S(X)/P_A + \omega N/P_A - F/P_A$$

In the absence of formal employment and an ICDP program, this constraint reads $C_A^c = A(E_A) - F/P_A$. This means that the local people must sell agricultural output to finance the imposed fine. However, in the presence of formal employment and an ICDP, the local people may purchase excess crops on the market as long as the additional income exceeds the imposed fine.

The next step is to present the resulting constraint on game meat consumption if caught in illegal hunting. In line with Skonhoft and Solstad (1998), it is assumed that the local people keep the illegal meat if caught. That is, they manage to hide the meat from the patrol units. In addition, as already mentioned, they will receive game meat from the culling program even if they are caught poaching. The resulting constraint on game meat consumption if caught therefore coincides with the prevailing constraint if they manage to escape in *(6)*, i.e. $C_G^c = C_G^e$.

Because the game meat consumption is independent of whether the local people are caught in illegal hunting, i.e. $C_G^c = C_G^e$, the expected utility function reads $EW = (1-\theta)U(C_A^e) + \theta U(C_A^c) + V(C_G)$, where the superscript on C_G is omitted. Substituting the consumption constraints *(6)*, *(7)* and *(9)*, together with constraints *(3)* and *(4)*, into this expression gives the expected utility function as

$$EW = (1-\delta E_h)\{U(A(E_A) + \mu S(X)/P_A + \omega(T - E_h - E_A)/P_A)\}$$
$$+ \delta E_h\{U(A(E_A) + \mu S(X)/P_A + \omega(T - E_h - E_A)/P_A - F/P_A)\} + V(f(E_h)X + mX).$$

The local people must decide upon the optimal effort use in hunting E_h and agricultural production E_A in order to maximize its expected utility. Because the local people treat the wildlife stock as exogenous, they impose no shadow price on wildlife. The Kuhn-Tucker first order maximum conditions are then given in *(10)-(11)*.

$$(10)\quad \left[(1-\delta E_h)U'(C_A^e)+\delta E_h U'(C_A^c)\right](A'(E_A)-\omega/P_A)\le 0\,;=0 \text{ if } E_A>0$$

$$(11)\quad V'(C_G)f'(E_h)X-\left[(1-\delta E_h)U'(C_A^e)+\delta E_h U'(C_A^c)\right]\omega/P_A$$
$$-\delta\left[U(C_A^e)-U(C_A^c)\right]\le 0\,;=0 \text{ if } E_h>0$$

Equation *(10)* gives the optimal effort use in agricultural production E_A^I, where superscript *I* denotes the case of Regime I. In what follows, an interior solution for E_A (> 0) is assumed to hold, so that the first order condition *(10)* holds with equality. The first parenthesis is positive and, hence, this condition reads $A'(E_A)-\omega/P_A=0$. This means that effort should be directed towards agricultural production until the value of the marginal product equals the wage rate in formal employment. Therefore, the agricultural productivity, the price of agricultural output, and the wage rate in formal sector are the only factors determining the optimal effort use in agricultural production. Hence, effort is allocated to agriculture independent of the amount of effort directed towards hunting and formal employment. This result is identical to what is demonstrated by Bulte and van Soest (1999).

Equation *(11)* gives the first order condition with respect to the hunting effort E_h^I. The first term on the left hand side reflects the marginal expected utility from hunting where more effort in illegal hunting increases the expected utility due to increased game meat consumption. The second and third terms give the marginal cost or disutility from hunting. The second term implies that more effort use in illegal hunting reduces the time in formal employment and, consequently, reduces the budget available for consumption of agricultural commodities. In addition, as seen in the third term, more hunting effort increases the probability of being caught, which reduces the expected utility for a given consumption bundle. The local people will refrain from illegal hunting, i.e. $E_h^I=0$, if the marginal disutility exceeds the marginal utility. This will be the case if the fraction of meat distributed to the community, the fine level, and the marginal probability of being detected are 'high'. However, because the intention of ICDP is to promote wildlife conservation by stimulating the local people to reduce the poaching, the case of no illegal hunting will not be considered in the following. Instead, it is assumed that an interior solution for E_h exists, where the local people divert effort to illegal hunting until the marginal utility of hunting equals the marginal disutility.

Having solved for E_A^I through *(10)*, E_h^I and X^I follow simultaneously in *(8)* (with *dX/dt* = 0) and *(11)*, while N^I is determined in *(4)*. The resulting consumption of game meat and the agricultural consumption, depending on whether the local people are caught in illegal hunting, follows from *(6)*, and *(7)* or *(9)*, respectively.

The next step is to analyse how the benefit-sharing instruments of ICDP influence the hunting decision of the local people, wildlife conservation and local welfare. The comparative static results in ecological equilibrium are derived from the derivatives of *(8)* (with *dX/dt = 0*) and *(11)*. The welfare effect follows from *(5)* when taking into account the impact of a changing wildlife stock[14]. See Appendix 1. Consider first the effect of an increase in the amount of game meat distributed from the managed culls to the local people, i.e. *m* increases. The direct effect on wildlife conservation is negative. The indirect effect works through a changing effort use in illegal hunting. Because there exists no external market where the local people can sell excess game meat, the only option is to consume the legal meat domestically (or within the community). The local people will therefore substitute illegal meat for distributed meat and reduce their effort in illegal hunting. Therefore, the indirect effect of game culling works in the direction of a higher degree of wildlife conservation. It can be demonstrated, however, that the direct connection is the dominating effect. Hence, while this policy fulfils the aim of reducing illegal hunting, the aggregate offtake increases and this lowers the degree of wildlife conservation. This result is in line with the findings of Barrett and Arcese (1998). By providing game meat to the local people, the management authority

[14] The actual welfare of the local people is strictly dependent on the realized consumption bundle, i.e. it is conditional upon whether they are caught in illegal hunting. More precisely, the actual welfare level is higher if the local people manage to escape the patrol units. However, we cannot observe which of the two states are realized. Instead, we investigate the welfare effect of ICDP by deriving its impact on the expected utility function.

contributes to increased pressure on the wildlife stock. The wildlife is therefore more protected without this kind of interference from the authorities.

Second, an increase in the income from tourism, i.e. μ increases, leads to an increase in illegal hunting and a smaller wildlife stock. The mechanism works as follows: money transfers which are received for certain increase the level of income in both of the states ‘escape’ and ‘caught’ (see section 2). This has two positive and quite similar effects on the hunting effort. First, it enables the local people to carry more risk. That is, they pay less attention to the probability of being caught when deciding upon the optimal use of effort in illegal hunting. Second, an increased certain income makes the local people less responsive to the fact that increased hunting effort reduces the income in formal employment. Both effects reduce the marginal cost of hunting and stimulate increased hunting effort. Consequently, certain income transfers reduce the degree of wildlife conservation. Although the mechanism is somehow different, this result is in accordance with both Barrett and Arcese (1998) and Skonhoft (1998), and suggests that ICDPs relying on money transfers fail to conserve wildlife.

Third, the effect of a positive shift in the wage rate ω in formal employment is generally unclear. All else equal, a higher wage level increases the certain income level. Quite similar to increased money transfers from tourism, this enables the local people to carry more risk and to draw back workers from the formal sector. Hence, this effect works in the direction of increased hunting effort. Compared to the money transfers from tourism, the difference lies in the fact that an increased wage rate in formal employment has a direct negative effect on hunting as it increases the alternative cost of effort use in this activity. The total effect is therefore unclear. However, if the latter effect dominates, then a higher wage rate will promote wildlife conservation. This will be the case if the marginal probability of detection is ‘high’ and the employment in formal sector is initially ‘low’. See also Table 1. In this case, a higher payment in formal sector will also improve the economic conditions of the local people. The conclusion is therefore that ICDPs relying on work opportunities in the formal sector may fulfil the aim of promoting both wildlife conservation and local welfare.

Fourth, improved agricultural productivity or a higher price of agricultural output P_A has an ambiguous effect on hunting effort and wildlife conservation. See Table 1. The effect is strictly dependent on the fine level. Recall from equation *(9)* that when the fine level is 'low', the local people will buy food on the market to supplement their consumption. In this case, the effect of a higher agricultural price is ambiguous. On the other hand, when faced with a 'high' fine level the local people will sell excess agricultural output on the market in order to finance the penalty if caught in illegal hunting. That is, the local people are net producers of agricultural output. Then, all else equal, a higher price increases the level of certain income which enables the local people to carry more risk and, therefore, increase the hunting effort. Hence, if the local people are net agricultural producers, policies which stimulate a higher agricultural return will lower the degree of wildlife conservation. This result is contrary to the arguments of Brown et al. (1993) and the findings of Skonhoft and Solstad (1998)[15].

The final policy option is to increase the degree of anti-poaching law enforcement in order to increase the marginal cost of illegal hunting. This policy includes more extensive use of guards and patrols, which increases the probability of being caught, and a higher fine level. Obviously, such attempts will promote wildlife conservation. However, the effect on the welfare of the local people is ambiguous. See Table 1. While the direct welfare effect is negative, there is a positive indirect effect working through a changing wildlife stock, as more wildlife increases the transfers of game meat and money from the tourism sector. If the latter effect dominates, law enforcement will promote both wildlife conservation and human welfare.

4. Regime II

The objective so far has been to investigate the impact on wildlife conservation and human welfare of the most common instruments of existing ICDPs. One of these instruments is benefit-sharing which consists of distribution of game meat and income

[15] Skonhoft and Solstad (1998) present a model of a producer (firm) who sells both agricultural output and game meat on the market, while no market exists for labour. In their model, the alternative cost of hunting equals the foregone return from agricultural production and, therefore, a higher agricultural price reduces the hunting effort. Assuming that $C_g < f(E_h)X + mX$, i.e. a market exists for game meat, and $N = 0$ makes the present model similar to the profit-maximizing model of Skonhoft and Solstad (1998).

transfers from tourism. Most existing ICDPs lack a proper link between the benefit transfers and illegal hunting, and section 3 demonstrated that transfers relying on this design do not have the potential of promoting wildlife conservation. Instead, it is clear that working opportunities in the formal sector provide, under given conditions, the most promising way of encouraging wildlife conservation and human welfare.

Despite the fact that game meat distribution and income transfers fail in meeting the aims of today's ICDPs, they are launched as having the potential to curtail illegal hunting and promote wildlife conservation. Therefore, the objective of this section is to look at an alternative design of the benefit-sharing strategies in order to reach the aim of integrated wildlife conservation and improved human welfare. In section 3 it was shown that a higher fine level reduces the illegal hunting pressure and promotes wildlife conservation. In addition, this policy may improve the economic conditions of the local people if the benefit transfers are 'high'. This suggests that one promising strategy may be to increase the costs of being caught in illegal hunting. One possible way is to attach an uncertainty to the benefit transfers so that participation in benefit sharing becomes conditioned by whether the local people are caught in illegal hunting. Then, in contrast to section 3, the transfers are no longer certain; the local people receive them if they manage to escape the patrol units, while they are denied transfers if they get caught. This section presents an ICDP design based on such a link between participation in benefit sharing and the imposed punishment if caught in illegal hunting.

Compared to Regime I, the present ICDP design restricts the resulting consumption possibilities of the local people if they are caught in illegal hunting. First, the local people receive no money transfers from tourism and, hence, $\mu = 0$. The resulting constraint on agricultural consumption yields

$$(12) \qquad C_A^c = A(E_A) + \omega N / P_A - F / P_A$$

Second, the local people are excluded from the game meat distribution program if they get caught in illegal hunting. Then, the constraint on game meat consumption equals[16]

$$(13) \qquad C_G^c = f(E_h)X$$

When inserting the consumption constraints in *(6)*, *(7)*, *(12)*, and *(13)*, together with the probability of being caught in equation *(3)* and the time constraint in *(4)*, the expected utility follows as $EW = (1-\delta E_h)\left[U((A(E_A)+\mu S(X)/P_A+\omega(T-E_A-E_h)/P_A)+V(f(E_h)X+mX)\right]$

$+\delta E_h\left[U((A(E_A)+\omega(T-E_A-E_h)/P_A-F/P_A)+V(f(E_h)X)\right]$. Again, the decision problem of the local people is to decide upon the optimal effort directed towards illegal hunting E_h and agricultural production E_A in order to maximize the expected utility. With an interior solution present (see section 3), the first order conditions for maximum are given in *(14)-(15)*.

$$(14) \qquad \left[(1-\delta E_h)U'(C_A^e)+\delta E_h U'(C_A^c)\right](A'(E_A)-\omega/P_A)=0$$

$$(15) \qquad \left[(1-\delta E_h)V'(C_G^e)+\delta E_h V'(C_G^c)\right]f'(E_h)X$$
$$-\left[(1-\delta E_h)U'(C_A^e)+\delta E_h U'(C_A^c)\right]\omega/P_A-\delta\left[U(C_A^e)+V(C_G^e)-U(C_A^c)-V(C_G^c)\right]=0$$

Equation *(14)* gives the optimal effort use in agricultural production E_A^{II}, where superscript *II* denotes Regime II. Again, the first parenthesis is positive, so that the first order condition reads $A'(E_A)-\omega/P_A=0$. Hence, the presence of a link does not alter the result that the optimal effort use in agriculture is determined by the price and productivity in agriculture and the wage rate in formal sector, but independent of the effort use in hunting and formal employment.

[16] The link presented here is implemented so that the management authority distributes money and meat at the end of a period, i.e. a quarter or a year. Then, the local people do not benefit if they have been caught in illegal hunting during that period.

Having solved for E_A^{II} in *(14)*, E_h^{II} and X^{II} follow simultaneously in *(15)* and *(8)* (with $dX/dt = 0$). The first order condition in *(15)* states that effort should be directed towards hunting until the marginal benefit (i.e. the first term) equals the marginal cost (i.e. the second and third term). For fixed parameter values, $(C_G^c)^{II} < (C_G^c)^I$ and $(C_A^c)^{II} < (C_A^c)^I$. Consequently, the first order condition in *(15)* differs in general from *(11)*. In order to compare the regimes, consider first the impact of a link on the income transfers from tourism. Because there is an uncertainty attached to the income transfers in Regime II, the realized consumption level of agricultural output if caught in illegal hunting is lower than in Regime I. Therefore, as seen from the second term in *(15)*, a link on the income transfers will increase the marginal cost of hunting and work in the direction of reduced hunting effort. There is an additional effect working through the fact that this link increases the gap between the consumption levels C_A^e and C_A^c. The resulting loss if being caught in illegal hunting is therefore higher in Regime II, which strengthens the negative impact on hunting effort. This means that ICDPs relying on a link between income transfers and the costs of being caught in illegal hunting will produce a higher degree of wildlife conservation.

Consider now the effect on illegal hunting of implementing a link on the game meat distribution. First, compared to Regime I the realized consumption level of game meat if caught is now lower. This increases the marginal expected utility of hunting and leads the local people to increase their hunting effort in order to compensate for the uncertainty of the transfers of game meat. The second effect works through the fact that this link increases the gap between consumption levels C_G^e and C_G^c. The loss resulting from being caught in illegal hunting is therefore higher in Regime II. Hence, this leads towards reduced hunting effort (see the third term in *(15)*). The total effect on hunting effort and wildlife conservation of a link on game meat distribution is therefore unclear. The numerical analysis in section 5 demonstrates under which conditions ICDPs relying on such a link will perform better than the ICDP design of today.

The next step is to investigate the effect on wildlife conservation and human welfare of increased transfers of game meat and income from the tourism sector. The comparative static results are derived from the derivatives of *(8)* (with $dX/dt = 0$), *(15)*, and *(5)* (see Appendix 1)[17]. First, consider the impact of increased game meat distribution m. In the same way as shown in section 3, the direct effect on hunting effort is negative as the local people substitute illegal meat for distributed meat in consumption. In the present scenario there is an additional negative effect working through the increased cost of being caught in illegal hunting. As a result, the effect on hunting effort is negative and stronger compared to Regime I. Increased distribution of game meat may therefore promote wildlife conservation. It can be demonstrated that the degree of wildlife conservation increases if game meat distribution is combined with policies that increase the probability of being caught in illegal hunting. See Table 1. If this is the case, culling operations may also succeed in promoting human welfare. The conclusion is therefore that transfers of game meat have the potential of encouraging both conservation and welfare if combined with anti-poaching law enforcement and if properly linked to the cost of being caught in illegal hunting.

The effect on hunting effort in Regime II of increased money transfers from tourism is ambiguous. The positive effect is the same as discussed for Regime I in section 3. However, creating a link between the money transfers and the involvement in illegal hunting increases the marginal cost of being caught. This additional effect works in the direction of reduced hunting effort. The total effect on illegal hunting and wildlife conservation is therefore unclear. However, as seen from Table 1, money transfers may promote wildlife conservation if combined with policies which increase the probability of being caught in illegal hunting. In this case, income transfers may also promote human welfare.

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[17] It is assumed that *(8)* reads $dX/dt = rX(1 - X/K) - f(E_h)X - mX$, also when the local people are caught in illegal hunting. This means that the park manager takes out a fraction m of the stock even if it is not distributed to the local people. Instead, the manager distributes the meat to the management staff or sells it on markets outside the region.

Table 1: The conservation effect of the respective policies and the corresponding welfare effect*.

	X^I	EW^I	X^{II}	EW^{II}
m	÷	+ if μ, m are 'small'; ÷ if μ, m are 'high'	+ if δ is 'high'; +/÷ otherwise	+ +/÷
μ	÷	+ if $S'(X)$, m are 'small'; ÷ if $S'(X)$, m are 'high'	+ if δ is 'high'; ÷ if δ is 'low';	+ ÷ if $S'(X)$, m are 'high'
ω	+ if δ is 'high', N is 'low'; +/÷ otherwise	+ +/÷	+ if δ is 'high', N is 'low'; +/÷ otherwise	+ +/÷
P_A	÷ if F is 'high'; +/÷ otherwise	+/÷	÷ if F is 'high'; +/÷ otherwise	+/÷
δ, F	+	+ if μ, m are 'high'; ÷ otherwise	+	+ if μ, m are 'high'; ÷ otherwise

* This table reports possible negative impact of m, μ and ω on human welfare. This occurs because of the effect working through a changing wildlife stock. If we assume that the local people are not able to calculate the impact on the future stock size, they may accept higher transfers and a higher wage level even if the welfare effect turns out as negative.

5. Numerical analysis

The theoretical reasoning will now be illustrated by data which fits the wildlife exploitation in the Serengeti-Mara ecosystem. This ecosystem is positioned on the border between Tanzania and Kenya and contains the world's largest ungulate herds (Sinclair and Arcese 1995, Barrett and Arcese 1998). The Serengeti National Park is a part of it, and compromises more than half of the ecosystem's land area (Barrett and Arcese 1998). The outer area of focus in the numerical analysis is the border area along the western corridor of the park where most of the poaching takes place. This area has experienced a rapid growth in human settlement (Campbell and Hofer 1995, Barrett and Arcese 1998) which coincides with a marked increase in the number of poachers arrested in the park (Arcese et al. 1995). As a result, Sinclair (1995, p. 24) states that "the illegal killing of the migrant ungulates by poachers is potentially the most serious threat to the Serengeti system".

The local people living on the western border of the park are mainly agro-pastoralists (Kauzeni and Kiwasila 1994). In addition, a survey conducted in Bunda and Serengeti District in 2001 predicts that almost 30 per cent of the households in this area are involved in illegal hunting (see Chapter 5 of this thesis). Hunting in the protected area is illegal, i.e. there are no local property rights to wildlife. However, local people in western Serengeti benefit from the existing ICDP in the area, namely the Serengeti Regional Conservation Project. This project was implemented during 1993/1994 and aims to improve wildlife conservation mainly through distribution of game meat from the managed harvest of wild ungulates (see Barrett and Arcese 1998 and Rugumayo 1999). In addition, a revenue-sharing programme exists for Robanda village in Serengeti District, under which the village receives money transfers from tourism activities established within the village area[18]. These benefit-sharing strategies are not subject to any risk of being expelled from the transfers as discussed in section 4. The current management regime in Serengeti is therefore characterised as an ICDP of Regime I.

The economic and ecological parts of the model are specified at the scale of one km^2 and one year. This means that the simulation results below report the wildlife density, that is, the number of animals per km^2. The closer definitions of the protected area and the outer area are found in Appendix 2. The baseline values for transfers, anti-poaching law enforcement, ecological data, and data for crop production and hunting used in the simulations are derived from the model of Regime I and also presented here. As demonstrated above, the conservation effect of money transfers and game meat distribution depend critically on the design of the benefit-sharing scheme. Because of the unclear effects, the coefficients m and μ, as well as ω, will be varied throughout the simulations.

[18] This revenue-sharing programme is of direct benefit to Robanda village. In addition the district authorities and the State gain revenues from fees paid by the tourism sector (Kauzeni and Kiwasila 1994), but the villages complain that they do not gain any income from these fees. For a broad overview of tourism activities in Serengeti, see Kauzeni and Kiwasila (1994). For the objectives of Tanzanian National Parks regarding revenue sharing in tourism, see TANAPA (1996).

Table 2 demonstrates how wildlife abundance varies with the culling coefficient *m* under the two regimes. The first column reports the results in Regime I, while the others give the results in Regime II when there is a link related to game meat distribution only (second column) and in the case where there also is a link to the income transfers from tourism (third column). In baseline $m = 0.0002$, and, consequently, wildlife density is $X^I = 50$ and $EW^I = 185$. As demonstrated in the theoretical analysis, increased legal offtake reduces the degree of wildlife conservation in this scenario. Compared to the baseline regime, the degree of wildlife conservation is barely affected by introducing a link between involvement in illegal hunting and the benefit transfers. This is because the current transfers generate such a small amount of legal meat and income so that, all else equal, the expected cost of being caught only just increases when a risk of being expelled from the benefit-sharing programme is created. On the other hand, we see that a 'high' culling rate is sufficient to ensure that a link on game meat distribution will promote both wildlife conservation and human welfare. In fact, this is the case for a culling rate up to 7 per cent of the wildlife stock. Hence, in this range, the reduction in illegal hunting more than offsets the legal offtake. Contrary to Barrett and Arcese (1998), this indicates that the culling programme may succeed in promoting both wildlife conservation and local welfare if the distribution of meat is properly linked to the illegal hunting. However, if the culling rate exceeds 7 per cent, the degree of conservation reduces with a higher culling rate.

Table 2: Simulation results of a changing culling fraction *m**.

	No risk of exclusion		Risk of exclusion from meat distribution		Risk of exclusion from meat and money transfers	
	X^I	EW^I	X^{II}	EW^{II}	X^{II}	EW^{II}
$m = 0.0002$	50	185	50	185	51	185
$m = 0.07$	35	187	57	191	57	191
$m = 0.09$	31	187	52	192	52	192

* Here, *X* measures the stock density. All parameters except *m* are fixed at their respective baseline value.

The simulations show, for both regimes, that the degree of wildlife conservation varies slightly with the money transfers from the tourism sector (see Appendix 2, Table A2).

This means that the risk of being excluded from the money transfers cannot conquer the effect working through an increased expected income. Consequently, as seen in Table 2, there is no additional conservation effect of linking both benefit transfers to the illegal hunting. This suggests that a link on game meat distribution combined with a higher culling rate is a more promising strategy in order to fulfil the aim of a higher degree of wildlife conservation and improved welfare for the local people.

Let us turn to the wage rate in the formal sector. The theoretical analysis of Regime I revealed an ambiguous relationship between wildlife conservation and the wage rate ω. The numerical analysis discloses, however, a positive relationship. See Table 3. This means that the increased alternative cost of hunting is the dominating effect. Consequently, subsidies which stimulate increased wage rate in formal employment will promote both wildlife conservation and human welfare.

Table 3: Simulation results of a changing wage rate ω*.

	X^I	EW^I
$\omega = 285$	50	185
$\omega = 430$	57	205
$\omega = 570$	60	229

* Here, X measures the stock density. All parameters except ω are fixed at their respective baseline value.

As discussed in the theoretical analysis of Regime I in section 3, a higher fine level may increase the degree of wildlife conservation without deteriorating the economic conditions of the local people. This will be the case if the conservation effect is relatively strong, so that the local community experiences a net gain due to increased transfers of game meat and money from the tourism sector. Table 4 demonstrates that the welfare is barely sensitive to a changing fine level. In fact, a double fine will increase the wildlife stock and leave local welfare unchanged.

Table 4: Simulation results of a changing fine level *F**.

	X^1	EW^1
$F = 110$	50	185
$F = 220$	54	185
$F = 440$	58	184

* Here, *X* measures the stock density. All parameters except F are fixed at their respective baseline value.

6. Discussion and conclusion

The attempt of wildlife ICDPs is to link conservation in protected areas to economic development in the surrounding communities. However, many of the existing ICDPs have experienced difficulties which may be traced to the specific design of the projects (see Brandon and Wells 1992, Wells and Brandon 1992, Barrett and Arcese 1995, Gibson and Marks 1995). The central contribution of this exercise is to highlight some possible pitfalls, and to clarify in what way the management design is crucial for the success of ICDPs. In order to do so, this paper presents a hunter-agrarian community located on the periphery of a protected area. Hunting performed by the local people is illegal, but the law enforcement imposed by the park manager is not sufficient to eliminate the illegal hunting. Markets exist for labour and agricultural commodities, while no market is present for game meat.

The theoretical model specifies two alternative ICDP designs for benefit transfers, i.e. distribution of game meat from managed culling and transfers of income from the tourism sector. In the first regime, the project manager fails to link the benefit transfers to the illegal hunting. Consequently, the local people receive game meat and money from tourism independent of whether they get caught in illegal hunting. This regime is in accordance with most of the existing ICDPs. In the second regime the project manager imposes on the local people a continuous risk of being expelled from the benefit transfers if caught in illegal hunting. Hence, in addition to the risk of receiving a monetary fine, there is also a risk of being denied benefit transfers.

It is demonstrated that the success of benefit sharing is conditional on the ICDP design. A benefit-sharing scheme implemented without a proper link to illegal hunting is less likely to succeed in gaining wildlife conservation. In fact, it turns out that both game meat distribution and money transfers from tourism will contribute to wildlife degradation in this regime. Transfers of game meat fail because the reduction in illegal hunting is not sufficient to offset increased culling. Money transfers from tourism fail because a higher level of the certain income enables the local people to carry more risk and makes them less dependent on the income from formal employment.

In order for benefit sharing to succeed, this analysis shows that there must be a risk for the local people of being expelled from the transfers if they get caught in illegal hunting. If such a risk is present, distribution of game meat and money transfers may succeed in promoting both wildlife conservation and human welfare. These results are in contrast to the conclusion of Barrett and Arcese (1998). In the case of Serengeti we have seen that a link on game meat distribution combined with a higher culling rate may lead to a higher degree of wildlife conservation and improved economic conditions of the local people.

Another important result of this study is that a higher return from formal employment may promote wildlife conservation. As long as the effect working through an increased alternative cost of hunting is relatively strong, the local people will shift the allocation of labour from illegal hunting to formal employment. This will be the case in areas with limited opportunities for formal employment and extensive use of anti-poaching law enforcement. For the case of Serengeti we have seen that a higher wage rate in formal employment reduces the pressure on wildlife and improves the livelihood of the local people.

The general conclusion of this analysis is that work should be done in order to design some type of explicit agreement over the benefit-sharing instruments between the management authorities and the local people. This agreement must specify the rights and duties of the respective parties and must be supported by enforceable penalties that provide enough incentives for the parties to comply. However, in practice, designing such contracts may be difficult, especially in poor African countries where the local people lack resources or power to secure their interests. Still, ICDP projects need to let go of the assumption that transfers and support alone will make people who live in periphery areas refrain from illegal hunting in absence of sufficient anti-poaching law enforcement and penalties (see also Wells and Brandon 1992). Projects partly depending on guard patrols and penalties are not inconsistent with the ICDP concept if combined with attempts to improve of human welfare.

References

Arcese, P., J. Hando and K. Campbell (1995): "Historical and present-day anti-poaching efforts in Serengeti". In Sinclair, A.R.E. and P. Arcese: *Serengeti II*, University of Chicago Press, Chicago, 506-533.

Barbier, E.B. (1992): "Community-based development in Africa". In Swanson, T.M. and E.B. Barbier (eds.): *Economics for the Wilds: Wildlife, Wildlands, Diversity and Development*, Earthscan, London, 103-135.

Barrett, C.B. and Arcese, P. (1995): "Are integrated conservation-development projects (ICDPs) sustainable? On the conservation of large mammals in Sub-Saharan Africa", *World Development*, 23(7), 1073-84.

Barrett, C.B. and P. Arcese (1998): "Wildlife harvest in integrated conservation and development projects: linking harvest to household demand, agricultural production, and environmental shocks in the Serengeti", *Land economics*, 74(4), 449-65.

Brandon, K. and M. Wells (1992): "Planning for people and parks: Design dilemmas", *World Development*, 20(4), 557-570.

Brown, K. et al. (1993): "Economics and the conservation of global biological diversity", *Global Environmental Facility, Working Paper.* No. 2, The World Bank, New York.

Bulte, E.H. and D. van Soest (1999): "A note on soil depth, failing markets and agricultural pricing", *Journal of Development Economics*, 58(1), 245-254.

Bulte, E. and G.C. van Kooten (1996): "A note on ivory trade and elephant conservation", *Environmental and Development Economics*, 1(4), 433-443.

Campbell, K. and H. Hofer (1995): "People and wildlife: Spatial dynamics and zones of interaction". In Sinclair, A.R.E. and P. Arcese: *Serengeti II*, University of Chicago Press, Chicago, 534-570.

Caughley, G. and A. Sinclair (1994): *Wildlife ecology and Management,* Oxford University Press, Oxford.

Dasgupta, P. (1993): *An Inquiry into Well-Being and Destitution*, Clarendon Press, Oxford.

Emerton, L. and I. Mfunda (1999): Making wildlife economically viable for communities living around the western Serengeti, Tanzania. *Evaluating Eden series Discussion Paper*, No. 1, International Institute for Environment and Development, London.

Gibson, C.C. and S.A. Marks (1995): "Transforming rural hunters into conservationists: An assessment of community-based wildlife management programs in Africa", *World Development*, 23(6), 941-57.

Hofer, et al. (2000): "Modeling the spatial distribution of the economic costs and benefits of illegal game meat hunting in the Serengeti", *Natural Resource Modeling*, 13(1), 151-177.

Kauzeni, A.S. and H.L. Kiwasila (1994): "Serengeti regional conservation strategy, a socio- economic study", Institute of Resource Assessment, University of Dar es Salaam, Tanzania.

Kiss, A. (1990): "Living with wildlife: Wildlife resources management with local participation in Africa." Technical Paper, 130. World Bank, Washington D.C.

Lopez, R. (1998): "Agricultural intensification, common property resources and the farm-household", *Environmental and Resource Economics*, 11(3-4), 443-458.

Marks, S.A. (1984): *The Imperial Lion: Human Dimensions of Wildlife Management in Africa*, Westview Press, Boulder.

Martin, R.B. (1993): "Should wildlife pay its way?" Department of national parks and wildlife management, Zimbabwe.

Milner-Gulland, E.J. and N. Leader-Williams (1992): "A model of incentives for the illegal exploitation of black rhinos and elephants: poaching pays in Luangwa Valley, Zambia", *Journal of Applied Ecology*, 29(2), 388-401.

Rugumayo, C.R. (1999): "The participation in sustainable resource management. The case of Serengeti regional conservation strategy, Tanzania", Working Paper 112, Norwegian Institute for Urban and Regional Research, Oslo.

Sadoulet, E. and A. de Janvry (1995): *Quantitative Development Policy Analysis*, The Johns Hopkins University Press, London.

Schulz, C.E. and A. Skonhoft (1996): "Wildlife management, land-use and conflicts", *Environment and Development Economics*, 1(3), 265-280.

Sinclair, A.R.E. (1995): Serengeti past and present, in Sinclair, A.R.E. and P. Arcese: *Serengeti II*, University of Chicago Press, Chicago, 3-30.

Sinclair, A.R.E. and P. Arcese (1995): Serengeti in the Context of Worldwide Conservation Efforts, in Sinclair, A.R.E. and Arcese, P.: *Serengeti II*, University of Chicago Press, Chicago, 31-46.

Skonhoft, A (1995): "On the conflicts of wildlife management in Africa", *International Journal of Sustainable Development and World Ecology*, 2, 267-277.

Skonhoft, A. (1998): "Resource utilization, property rights and welfare - Wildlife and the local people", *Ecological Economics*, 26(1), 67-80.

Skonhoft, A. (1999): "On the optimal exploitation of terrestrial animal species", *Environmental and Resource Economics*, 13(1), 45-57.

Skonhoft, A. and J.T. Solstad (1998): "The political economy of wildlife exploitation", *Land Economics*, 74(1), 16-31.

Smith, V.L. (1975): "The primitive hunter culture, pleistocene extinction, and the rise of agriculture", *Journal of Political Economy*, 83(4), 727-755.

Songorwa, A.N. (1999): "Community-based wildlife management (CWM) in Tanzania: Are the communities interested?", *World Development*, 27(12), 2061-79.

SRCP (1998): "Overview of the communities surrounding the Ikorongo/Grumeti Game Reserves", Report for the Ministry of Natural Resources and Tourism, Arusha, Tanzania.

TANAPA (1996): "Serengeti national park. Management zone plan/environmental impact assessment", Tanzanian National Parks, Tanzania.

Wells, M. and K. Brandon (1992): *People and Parks: Linking Protected Area Management with Local Communities*, World Bank, World Wildlife Fund and U.S. Agency for International Development, Washington DC.

Wright, R. (1999): "Wildlife management in the national parks: questions is search of answers", *Ecological Applications*, 9, 30-36.

Appendix 1

1. Regime I

In Regime I, where there is no link between benefit transfers and involvement in illegal hunting, the effect on hunting effort and wildlife conservation of altering the management instruments are found by taking the total differential of *(8)* (with dX/dt = 0) and *(11)*. In the following, we specify $U(C_A) = k_A C_A^{\alpha}$ and $V(C_G) = k_G C_G^{\beta}$, where $k_i > 0$ $i = A, G$, $0 < \alpha \le 1$ and $0 < \beta \le 1$. For risk-averse poachers $\alpha < 1$ and $\beta < 1$. The differential is given in *(A1)* where λ_{E_h} denotes the derivative of *(11)* with respect to E_h, λ_X the derivative of *(11)* with respect to X etc.

$$\begin{bmatrix} 1 & f'(E_h^I)K/r \\ \lambda_X & \lambda_{E_h} \end{bmatrix} \begin{bmatrix} dX^I \\ dE_h^I \end{bmatrix} = \begin{bmatrix} -K/r \\ \lambda_m \end{bmatrix} dm + \begin{bmatrix} 0 \\ \lambda_\mu \end{bmatrix} d\mu$$

(A1)

$$+\begin{bmatrix} 0 \\ \lambda_\omega \end{bmatrix} d\omega + \begin{bmatrix} 0 \\ \lambda_{P_A} \end{bmatrix} dP_A + \begin{bmatrix} 0 \\ \lambda_\delta \end{bmatrix} d\delta + \begin{bmatrix} 0 \\ \lambda_F \end{bmatrix} dF$$

The sign of $\lambda_{E_h} = \beta(\beta-1)k_G C_G^{\beta-2}\left[f'(E_h)X\right]^2$ $+\beta k_G C_G^{\beta-1} f''(E_h)X + \alpha(\alpha-1)k_A (C_A^e)^{\alpha-2}(\omega/P_A)^2 + 2\delta\alpha\left[(C_A^e)^{\alpha-1} - (C_A^c)^{\alpha-1}\right]\omega/P_A$ is negative, while the sign of

$$\lambda_X = \beta^2 k_G C_G^{\beta-1} f'(E_h) - \alpha(\alpha-1)k_A C_A^{\alpha-2}\omega\mu S'(X)/P_A^2 - \delta\alpha k_A\left[(C_A^e)^{\alpha-1} - (C_A^c)^{\alpha-1}\right]\mu S'(X)/P_A$$

is positive. The determinant of the system, $\lambda_{E_h} - \lambda_X f'(E_h)K/r$, is therefore negative.

$\lambda_m = \beta(1-\beta)k_G C_G^{\beta-2} f'(E_h)X^2 > 0$,

$\lambda_\mu = \alpha(\alpha-1)k_A\left[(1-\theta)(C_A^e)^{\alpha-2} + \theta(C_A^c)^{\alpha-2}\right]\omega S(X)/P_A$ $+\delta\alpha k_A\left[(C_A^e)^{\alpha-1} - (C_A^c)^{\alpha-1}\right]S(X)/P_A < 0$. The signs of

$\lambda_\omega = \alpha(\alpha-1)k_A\left[(1-\theta)(C_A^e)^{\alpha-2} + \theta(C_A^c)^{\alpha-2}\right]\omega N/P_A^2$ $+\alpha k_A\left\{\left[(1-\delta(E_h-N)\right](C_A^e)^{\alpha-1} + \delta(E_h-N)(C_A^c)^{\alpha-1}\right\}/P_A$ and

$\lambda_{P_A} = \alpha(\alpha-1)k_A\left[(1-\theta)(C_A^e)^{\alpha-2}(A(E_A) - C_A^e) + \theta(C_A^c)^{\alpha-2}(A(E_A) - C_A^c)\right]\omega/P_A^2$

$-\alpha k_A\left[(1-\theta)(C_A^e)^{\alpha-1}+\theta(C_A^c)^{\alpha-1}\right]\omega / P_A^2$

$+\delta\alpha k_A\left[(C_A^e)^{\alpha-1}(A(E_A)-C_A^e)-(C_A^c)^{\alpha-1}(A(E_A)-C_A^c)\right]/ P_A$ are unclear. Finally, the signs of $\lambda_\delta=-E_h k_A \alpha\left[(C_A^e)^{\alpha-1}-(C_A^c)^{\alpha-1}\right]\omega / P_A+k_A\left[(C_A^e)^\alpha-(C_A^c)^\alpha\right]$ and $\lambda_F=-\theta\alpha(\alpha-1)k_A(C_A^c)^{\alpha-2}\omega / P_A^2+\delta\alpha k_A(C_A^c)^{\alpha-1}/ P_A$ are positive.

The welfare effect

The effect on local welfare is derived by taking the differential of *(5)* when taking into account the effect of a changing wildlife stock in *(8)* as

$$
\begin{aligned}
\text{(A2)}\quad dEW = & -\left\{\alpha k_A\left[(1-\theta)(C_A^e)^{\alpha-1}+\theta(C_A^c)^{\alpha-1}\right]\mu S'(X)/P_A \right. \\
& \left. +\beta k_G C_G{}^{\beta-1}(f(E_h)+m)\right\}(f'(E_h)K/r)dE_h \\
& +\left\{\beta k_G(C_G)^{\beta-1}X-\alpha k_A\left[(1-\theta)(C_A^e)^{\alpha-1}+\theta(C_A^c)^{\alpha-1}\right]\mu S'(X)K/rP_A \right. \\
& \left. -\beta k_G(C_G)^{\beta-1}(f(E_h)+m)K/r\right\}dm \\
& +\alpha k_A\left[(1-\theta)(C_A^e)^{\alpha-1}+\theta(C_A^c)^{\alpha-1}\right](S(X)/P_A)d\mu \\
& +\alpha k_A\left[(1-\theta)(C_A^e)^{\alpha-1}+\theta(C_A^c)^{\alpha-1}\right](N/P_A)d\omega \\
& +(\alpha k_A/P_A^2)\left[(1-\theta)(A(E_A)-C_A^e)^{\alpha-1}+\theta(A(E_A)-C_A^c)^{\alpha-1}\right]dP_A \\
& -E_h k_A\left[(C_A^e)^\alpha-(C_A^c)^\alpha\right]d\delta-(\theta\alpha k_A(C_A^c)^{\alpha-1}/P_A)dF
\end{aligned}
$$

2. Regime II

In the present of a link the effects are found by taking the total differential of *(8)* (with *dX/dt* = 0) and *(15)* as in *(A3)*.

$$
\text{(A3)}\quad \begin{bmatrix} 1 & f'(E_h)K/r \\ \lambda_X & \lambda_{E_h} \end{bmatrix}\begin{bmatrix} dX \\ dE_h \end{bmatrix}=\begin{bmatrix} -K/r \\ \lambda_m \end{bmatrix}dm+\begin{bmatrix} 0 \\ \lambda_\mu \end{bmatrix}d\mu
$$

Again, we specify the utility function as $U=k_A C_A^\alpha+k_G C_G^\beta$, where $k_i>0$ $i = A, G$, $0<\alpha\le 1$ and $0<\beta\le 1$. The sign of

$\lambda_{E_h}=\beta(\beta-1)k_G\left[(1-\theta)(C_G^e)^{\beta-2}+\theta(C_G^c)^{\beta-2}\right]\left[f'(E_h)X\right]^2$

$+\beta k_G\left[(1-\theta)(C_G^e)^{\beta-1}+\theta(C_G^c)^{\beta-1}\right]f''(E_h)X-2\delta\alpha k_A\left[(C_A^e)^{\alpha-1}-(C_A^c)^{\alpha-1}\right]\omega / P_A$

$-2\delta\beta k_G\left[(C_G^e)^{\beta-1}-(C_G^c)^{\beta-1}\right]f'(E_h)X$ is negative from the second order maximum condition. It is assumed that the sign of

$\lambda_X = f'(E_h)\beta^2 k_G\left[(1-\theta)(C_G^e)^{\beta-1}+\theta(C_G^c)^{\beta-1}\right]+\alpha\mu S'(X)(C_A^e)^{\alpha-2}k_A\left[(1-\theta)(1-\alpha)\omega/P_A-C_A^e\right]$

$-\delta\beta k_G\left[f(E_h)((C_G^e)^{\beta-1}-(C_G^c)^{\beta-1})+m(C_G^e)^{\beta-1}\right]$ is positive, which holds whenever μ and m are 'not too high'. Then, the determinant of the system, $\lambda_{E_h}-\lambda_X f'(E_h)K/r$, is negative. The sign of $\lambda_m=(1-\theta)\beta(1-\beta)k_G(C_G^e)^{\beta-2}f'(E_h)X^2+\delta\beta k_G(C_G^e)^{\beta-1}X$ is positive, while the sign of $\lambda_\mu=\alpha k_A(C_A^e)^{\alpha-2}\left[(1-\theta)(\alpha-1)\omega/P_A+\delta C_A^e\right]S(X)/P_A$ is in general unclear.

The welfare effect

Again, the effect on local welfare is derived by taking the differential of *(5)* when taking into account the effect of a changing wildlife stock in *(8)* as

(A4)
$$
\begin{aligned}
dEW = &-\{\alpha(1-\theta)k_A(C_A^e)^{\alpha-1}\mu S'(X)/P_A \\
&+\beta k_G\left[(1-\theta)(C_G^e)^{\beta-1}(f(E_h)+m)+\theta(C_G^c)^{\beta-1}f(E_h)\right]\}(f'(E_h)K/r)dE_h \\
&+\{\beta(1-\theta)(C_G^e)^{\beta-1}-\alpha(1-\theta)k_A(C_A^e)^{\alpha-1}\mu S'(X)K/rP_A \\
&-\beta k_G\left[(1-\theta)(C_G^e)^{\beta-1}(f(E_h)+m)+\theta(C_G^c)^{\beta-1}f(E_h)K/r\right]\}dm \\
&+\alpha(1-\theta)k_A(C_A^e)^{\alpha-1}(S(X)/P_A)d\mu
\end{aligned}
$$

Appendix 2

The numerical analysis

As mentioned in the main text, the ecological model is specified for the numerical analysis at the scale of one km^2 and one year. The same scale is also used for the agricultural benefit, as well as the hunting benefit, given in 1998/99 prices. The protected area, consisting of Serengeti National Park (SNP) and its surrounding game reserves, covers an area of some 26 000 km^2 (TANAPA 1996). The 'outer area' is thought of as the surrounding area on the western edge of the protected land. Campbell and Hofer (1995) identify the catchment area, i.e. the region in which the poachers reside, as the area within a maximum distance of 45 km to the protected land. This region constitutes some 30 500 km^2 and is, in this numerical analysis, interpreted as the 'outer area'. The human population in this region is estimated to be about 1.1 million with an average household size of about 7 persons (Campbell and Hofer 1995). Accordingly, there will be about 5 households per km^2 in the outer area. On average, it is assumed that 2 persons per household work in agricultural production, hunting, and formal sector. Hence, the effort constraint T is 10 man-labour years and, hence, $10 = E_A + E_h + N$.

The numerical analysis is exemplified by the wildebeest exploitation. The wildebeest population is estimated to be about 1.3 million animals and the annual offtake to some 120 000 animals (Campbell and Hofer 1995). The wildlife density in the protected area is therefore 50 animals per km^2, while the offtake is some 5 animals per km^2. Following Campbell and Hofer (1995), it is assumed to be 0.2 hunters per average household in western Serengeti. Consequently, it is one person involved in hunting at full time basis for every 5 households, and the baseline value of E_h is accordingly 1. The hunting function in *(2)* is specified as $h = qE_h^{\gamma} X$, where q is the catchability coefficient and γ is a scale parameter. γ is set to 0.9 (Barrett and Arcese 1998). By imposing the baseline value of E_h into the hunting function with $h / X = 5/50$, q is calculated to 0.1. The baseline value of the legal offtake is calculated from the hunting quotas of SRCP for the year 2000 hunting season. Based on a quota of 15 wildebeest per project village, m is set to 0.0002. The maximum specific growth rate is fixed as r = 0.3 (Caughley and

Sinclar 1994). To calculate the wildlife stock at its base level the carrying capacity *K* is set to 75 animals per km^2, meaning that the protected area can carry a stock of wildebeest just below 2 million animals.

The agricultural yield function is specified as $A(E_A) = z(E_A)^{\sigma}$ with *z* as a productivity parameter and σ as a scale parameter. The scale parameter is given as 0.8 (Barrett and Arcese 1998). According to a questionnaire among 300 households in Serengeti and Bunda Districts in 2001 (see Chapter 5 of this thesis), the average plot size per household is 7.4 acres, corresponding to a cultivated area of a fraction of 0.15 per km^2 for the average 5 households. For the same districts, the value of the crop production is estimated to US$ 5 861 000 or some US$ 19 000 per km^2 cultivated land (Emerton and Mfunda 1999). At our scale of one km^2, this represents a value of US$ 19 000×0.15 = US$ 2 850 (or US$ 570 per household). This is assumed to be representative for the whole outer area. The main crops grown in western Serengeti are sorghum, cassava, maize, and cotton (SRCP 1998). Personal communication with SRCP (1999) indicates a per kg price of US$ 0.18 for sorghum, US$ 0.05 for cassava, US$ 0.11 for maize, and US$ 0.19 for cotton. By weighting the crop prices by the relative magnitude of these crops (SRCP 1998), the price per kilo agricultural output equals US$ 0.15, so that P_A = 0.15. The time constraint in *(3)* gives $E_A = T - E_h - N = 10 - 1 - N$. Because a large fraction of the households in western Serengeti lack the opportunity of formal employment, it is assumed that only 20 per cent of the households have one person employed at full time basis in the formal sector. This means that the baseline value of *N* is set to 1 and, hence, E_A to 8. Consequently, the value of the crop production, i.e. $P_A z E_A{}^{0.8} = 2850$, is balanced with $z = 3600$. This means that one labour year in agricultural production gives an output of 3600 kilos crops.

The wage rate in formal employment follows from the first order condition in *(10)* which balances with $\omega = 285$, i.e. the annual income of full-time employment is US$ 285. This corresponds well with the average wage of US$ 0.8 per day paid in the food processing industry in Mara Region (Hofer et al. 2000). The income from tourism *S(X)* is interpreted as the revenue from public fees (entering fees, bed fees etc.) paid by tourists visiting SNP. According to Kauzeni and Kiwasila (1994), the income from fees in 1993 was US$ 420 000 or some US$ 5 per tourist. It is assumed that the average fee is fixed at the level of US$ 5. Kauzeni and Kiwasila (1994) report that the number of tourists visiting SNP in 1990 was 63 000 with an average annual growth equal to 7000 tourists in the period of 1984-1990. Using the same annual growth, the number of visitor in 1999 is calculated to 126 000. This gives an income from fees of US$ 630 000. Records from the village administration in Robanda show that this village received US$ 17 000 from the wildlife lodge in its village area. This corresponds to 3% of the annual tourism income and, hence, μ is set to 0.03. Because the model is specified at the scale of one km^2, we must correct for this in *S(X)*. The ratio of the tourism income to the value of crop production equals 0.11 which must also be the case at the scale of one km^2. Therefore, the baseline value of *S(X)* is set to 314. *S(X)* is specified as $P_T \varepsilon \ln X$, where $P_T = 5$ is the average fee paid per tourist and $\varepsilon \ln X$ is the number of tourists which depends on the wildlife density, where $\varepsilon > 0$. Then, solving for ε gives $\varepsilon = 16$. When it comes to the probability of being caught in illegal hunting, Hofer et al. (2000) provide an estimate equal to 0.002 per day. If the hunter spends all hunting effort on one continuously hunting trip, then the probability of being caught equals $0.002 \times 365 = 0.7$. In reality, however, the hunter divides the hunting effort between several hunting trips lasting for a number of days, where the probability of being caught on one particular trip may be independent of past and future trips. In this case, a value of 0.7 represents an overestimation of δ. In the following, we set the baseline of δ to half of this value, i.e. $\delta = 0.35$. This means that the probability of being caught for a full time hunter is 0.35 a year. Based on Hofer et al. (2000), the fine F equals US$ 110.26. Finally, $U(C_A)$ is specified as $U = k_A C_A{}^{\alpha}$ with $\alpha = 0.5$ and $k_A = 1$, while $V(C_G)$ is specified as $V = k_G C_G{}^{\beta}$ with $\beta = 0.2$. Then, in order to fit the model to its baseline values k_G is set to 30. Table A1 summarises the baseline parameter values.

Table A1: Baseline values economical and ecological parameters

Parameter	Description	Value
P_A	Crop price	0.15 ($/kg)
α	Scale parameter utility of agric. output	0.5
k_A	Linear parameter utility of agric. output	1
β	Scale parameter utility of game meat	0.2
k_G	Linear parameter utility of game meat	30
σ	Input elasticity labour crop production	0.8
z	Productivity crop production	3600 (kg)
T	Available labour effort, man-labour years	10
q	Catchability coefficient	0.1
γ	Input elasticity labour hunting	0.9
ω	Wage rate formal employment	285 ($)
μ	Fraction of tourism income transferred to every 5 households	0.03
P_T	Average fee	5 ($/tourist)
ε	Constant, tourism income	16
δ	Probability of detection in illegal hunting	0.35
F	Fine imposed if detected in illegal hunting	110.26 ($)
T	Total available effort	10 (labour year)
m	Cropping ratio	0.0002
K	Carrying capacity	75 (animal/km^2)
r	Intrinsic growth rate	0.3

Table A2: Simulation results of changing the money transfers μ*.
In Regime II: link on money transfers only.

	X^I	EW^I	X^{II}	EW^{II}
$\mu = 0.03$	50	185	51	185
$\mu = 0.15$	50	186	52	191

* Here X measures the stock density. All parameters except μ are fixed at their respective baseline value.

Chapter 5

Wildlife conservation policies and incentives to hunt:
An empirical analysis of illegal hunting in western Serengeti, Tanzania

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Abstract:
This paper estimates functions for effort use in illegal hunting using cross-sectional survey data from households in western Serengeti, Tanzania. One purpose of the analysis is to investigate the impact on illegal hunting of the integrated conservation and development project established in this area, namely the Serengeti Regional Conservation Project (SRCP). We also investigate how the pattern of crop production in agriculture and wildlife-induced damage to crops and domestic animals affect illegal hunting. The empirical results show that the hunting effort is inversely related to participation in SRCP and positively related to the degree of crop production for own consumption. However, the data indicates that the SRCP of today cannot improve the economic conditions in the project villages. Instead, in order to promote both wildlife conservation and human welfare, policymakers should encourage a higher degree of crop production for the market. In addition, assistance and support for preventing wildlife-induced damage have the potential of reducing the hunting pressure and improving the economic conditions in agriculture.

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1. Introduction

Protected areas such as national parks and game reserves have long been regarded as crucial in wildlife conservation. The effort to restrict the impact of humans in protected areas has traditionally been concentrated around guard patrols and penalties in order to discourage and prevent encroachment and illegal activities. However, during the past decades, this kind of exclusionary protected area management has been viewed as having failed to preserve wildlife in developing countries (Kiss 1990, Barrett and Arcese 1995, Gibson and Marks 1995). Today there is a growing recognition that the successful management of protected areas depends on the co-operation and support of the local people living with wildlife. In response to this, Integrated Conservation and Development Projects (ICDPs) are frequently adopted in developing countries. These projects aim at changing the incentives of rural inhabitants through benefit-sharing schemes (e.g. direct utilization of wildlife, income transfers from the tourism sector, improved conditions in the agricultural sector etc.), awareness building, and education in wildlife conservation.

ICDPs are based on the assumption that an appropriate set of incentives exists to induce people to change their exploitation practices. However, these projects have recently been the focus of attention because of the untested assumptions behind their strategies. Based on case studies of existing ICDPs, Brandon and Wells (1992) discuss design dilemmas and highlight possible pitfalls regarding their benefit-sharing schemes. They point to several issues that make the design complex, such as defining what uses to allow and how to link the benefit-sharing approach with the conservation objective. Benefit-sharing strategies have also been of focus in the bio-economic literature. For instance, Barrett and Arcese (1998) reveal possible undesirable effects of culling programmes where game meat is distributed to the local people, while Lopez (1998) and Bulte and van Soest (1999) show that improved agricultural productivity may result in increased resource exploitation.

A common feature in the bio-economic models referred to above is that they consider the incompleteness or absence of particular markets. This is a structural characteristic of rural areas in developing countries (de Janvry et al. 1991). Access to markets in remote communities tends to be limited by the large transaction costs associated with geographic isolation and poor infrastructure. Some products are therefore likely to be selected for subsistence use rather than for sale in small towns or other regional markets.

In a household model with no market for game meat, Barrett and Arcese (1998) analyse the wildebeest exploitation in western Serengeti in Tanzania. The existing ICDP in this area is the Serengeti Regional Conservation Project which, among other strategies, distributes game meat to the local people from managed culling operations. Barrett and Arcese investigate the impact of this strategy in a sensitive analysis where the household derives utility from consumption of game meat, agricultural output and leisure. This model implies that increased endowment of game meat through a managed game meat distribution programme reduces the illegal offtake. However, because game meat consumption is a normal good, there is a less than one-for-one trade-off of distributed meat for illegal meat, meaning that the aggregate wildlife offtake increases[1]. Hence, game meat distribution reduces the degree of wildlife conservation.

Another strategy repeatedly proposed in order to promote wildlife conservation is to implement policies which improve the economic conditions in the agricultural sector. For instance, Brown et al. (1993) suggest that improved productivity of labour in agriculture will divert labour away from hunting and thereby reduce the pressure on wildlife. Productivity improvements may be attained through the support of more sophisticated technologies, expansion of irrigation systems, more extensive fertilizer and pesticide use, and easier access to land. Lopez (1998) and Bulte and van Soest (1999) investigate the impact of productivity improvements with varying assumptions about the presence of markets. Bulte and van Soest (1999) assume that there exists a market for agricultural output and demonstrate that the impact on wildlife conservation of increased agricultural productivity is critically dependent on whether there exist markets for game meat and labour. With such markets present, the household solves for

[1] This result follows automatically from the assumption that game meat is non-tradable, i.e. the consumption C equals the sum of illegal meat h and distributed meat M as $C = h + M$. Partial derivation of this constraint with respect to M gives $\partial C / \partial M = \partial h / \partial M + 1$. Because game meat is a normal good, i.e. $\partial C / \partial M > 0$, it follows from this equality that $\partial h / \partial M > -1$.

the optimal labour use in agriculture and hunting separately[2]. Consequently, improved productivity in the agricultural sector has no effect on wildlife exploitation. With no markets present, they show that the conservation effect of productivity improvements is ambiguous.

Lopez (1998) presents a model where the household produces two agricultural commodities, a labour-intensive one and a land-intensive one. He investigates the impact of productivity improvements in agriculture when the household participates in the markets for agricultural commodities and the extracted resource. In the case of fixed endowment of land and no market for labour, he shows that increased productivity of the labour-intensive output is likely to increase the labour demand for farming, and hence reduce the resource extraction. In contrast, increased productivity of the land-intensive output results in increased resource exploitation.

The bio-economic analyses referred to above use theoretical models and numerical analyses in order to identify the impact on resource exploitation of game culling and productivity improvements in agriculture. Moreover, few, if any, empirical regression analyses of the economic incentives to hunt exist in the literature. The present paper adds to the scarce empirical literature by undertaking an analysis of the hunting incentives using cross-sectional survey data from households in western Serengeti. The survey was conducted in the period June to August 2001 among the local communities along the western border of the Serengeti National Park. This area has experienced a rapid growth in human settlement (Campbell and Hofer 1995, Barrett and Arcese 1998) which coincides with a marked increase in the number of poachers arrested in the park (Arcese et al. 1995). Today, Serengeti National Park and its surrounding game reserves contain the world's largest ungulate herds (Sinclair and Arcese 1995, Barrett and Arcese 1998), but Sinclair (1995, page 24) states that "the illegal killing of the migrant ungulates by poachers is potentially the most serious threat to the Serengeti ecosystem".

[2] The intuition is that when a labour market is present (e.g. formal employment) the household is able to alter the effort use in agriculture by adjusting its labour supply in formal employment, while the effort use in wildlife harvesting is left unchanged, and vice versa.

The local people in Serengeti have no legal rights to exploit wildlife, but hunt illegally[3]. The purpose of this paper is to examine what may induce the local people to participate in this activity. We investigate the effect of the Serengeti Regional Conservation Project (SRCP) which was implemented in 1993/94 as a response to the increasing pressure from the communities on the western border of Serengeti National Park. The intention of the project is to reconcile wildlife conservation and development for the human populations by providing wildlife benefits to the local people. The main strategy of SRCP is to distribute game meat to the project villages (see SRCS 1993, SRCS 1995, Rugumayo 1999). SRCP has also assisted the training of village game scouts and the establishment village wildlife funds (see also section 3.1).

In addition to the impact of SRCP, this paper investigates how the illegal hunting is related to the crop composition in agriculture. Here, we distinguish between cash crops and food crops sold on the market, and food crops produced for own consumption only. This distinction allows us to analyse the impact of access and participation in the market for agricultural crops on illegal hunting. The third contribution of this paper is to investigate how wildlife-induced damage to crops and domestic animals affects illegal hunting.

The rest of the paper is organized as follows. Section 2 presents the theoretical modelling of the hunting decision of the household. The data set is presented in section 3, while the empirical specification and the estimation results are derived in section 4. Finally, section 5 contains a summary and discussion of the main findings in the paper.

2. The theoretical model

The starting point of the theoretical modelling is an ecosystem that serves as a habitat for wildlife and a living area for humans. The local people are involved in two production activities; agricultural crop production and wildlife harvesting. The State or a management authority has the property rights to wildlife and wildlife utilization

[3] Hunting is not strictly illegal in parts of the western side of Serengeti National Park and persons having a licence are allowed to hunt wildlife. However, none of the respondents in this survey have such a licence and, hence, all hunting reported is illegal.

performed by the local people is illegal. Still, illegal hunting takes place due to lack of anti-poaching law enforcement. There are two basic motives behind hunting. First, the local people hunt to supplement domestic income and game meat consumption. In addition, they hunt to get rid of 'problem' animals destroying agricultural crops.

The following investigates the impact on illegal hunting of game meat transfers to the local people from managed culling operations, productivity improvements in agriculture, and policies aimed at reducing the wildlife-induced damage. The theoretical model also captures the effect of income transfers from the tourism sector, a tool which is frequently implemented in existing ICDPs. It is demonstrated that such transfers may provide the household with sufficient money to purchase game meat on the market. It turns out that this affects the impact on illegal hunting of a changing price of game meat.

Assume that the local people constitute a group of *n* identical households. Following de Janvry et al. (1991), the representative household diverts labour between agricultural crop production and illegal hunting, so as to maximize the utility it derives from consumption of these goods. The utility function is given by

$$(1) \qquad U^i = U(C_c^i, C_m^i), \qquad\qquad i = 1,\dots,n$$

where C_c^i is the consumption of agricultural crops and C_m^i is the consumption of game meat in household *i*. The utility function is assumed to have the regular properties, i.e. strictly quasi-concave with positive marginal utilities.

The agricultural crop production depends on effort use E_c^i and the amount of agricultural land L^i, while we disregard the use of fertilizer, pesticides etc. In the same way as Barrett and Arcese (1998), one of Lopez's models (1998), and Bulte and van Soest (1999), the land cultivated for crops is of fixed size. The production function is given as

(2) $$R^i = f(E_c^i; L^i),$$

where output increases by a decreasing rate in effort, $f_1 > 0$ and $f_{11} \leq 0$. Wildlife roaming in the village area causes damage to agricultural production. Following Zivin et al. (2000) it is assumed that the damage increases with the size of the wildlife stock *X*. The fraction of agricultural crops in household *i* destroyed by wildlife is $DC^i = DC^i(X)$ where $DC^i \in [0,1]$ and $dDC^i / dX > 0$ (see also Carlson and Wetzstein 1994). The net agricultural output is therefore given by $R^i[1 - DC^i(X)]$.

Game meat is produced through illegal hunting of wildlife. The wildlife harvesting function of household *i* is given by the Schaefer function as

(2) $$h^i = qE_h^i X,$$

where E_h^i is effort directed to hunting and *q* is the catchability coefficient identical for all household. Let M^i be the fixed endowment of effort in household *i*. The constraint on labour use reads

(3) $$M^i = E_c^i + E_h^i$$

An alternative cost of effort use in hunting is therefore present.

It is assumed that the household consumption of agricultural crops is constrained by the net production as given in *(4)*. The household sells excess crops on the market whenever the domestic consumption is below the level of crop production. In this case, the constraint in *(4)* is non-binding. Otherwise, the household produces crops for own consumption only, which means that the constraint is binding.

(4) $$C_c^i \leq f(E_c^i; L^i)[1 - DC^i(X)]$$

The household maximizes its utility given the time constraint *(3)*, the constraint on crop consumption in *(4)*, and a budget constraint. The budget constraint is outlined as follows. The consumption of game meat in household i consists of illegally harvested game meat h^i, and the legal game meat S^i distributed from the management authorities. Following Barrett and Arcese (1998), S^i is fixed and distributed freely to the household. Illegal and legal meat are considered homogenous in consumption and sold at the same price P_m on the market. Z^i is a fixed composed factor of labour-free income (e.g. money transfers from tourism), taxes and costs related to the purchase of basis goods (clothes, housing etc.). It is assumed that $Z^i > 0$ if the household receive 'high' transfers from tourism, while $Z^i < 0$ reflects 'high' fixed costs and taxes. Let P_c be the unit price of agricultural crops. Then, the household faces the following budget constraint[4].

$$
\begin{aligned}
&P_c C_c^i + P_m C_m^i \\
(5)\qquad & \\
&= P_c f(E_c^i; L^i)\left[1 - DC^i(X)\right] + P_m\left[qE_h^i X + S^i\right] + Z^i
\end{aligned}
$$

If $Z^i < 0$ and the constraint on crop consumption is binding, the household is a net producer of meat, i.e. $C_m^i < qE_h^i X + S^i$. This is also the case when the constraint on crop consumption is non-binding combined with a 'high' fixed cost Z^i (< 0). In this case, the household must sell excess meat on the market in order to finance the fixed cost. On the other hand, 'high' money transfers from tourism, i.e. Z^i is positive, and a non-binding constraint on crop consumption must be offset by net consumption of game meat, i.e. $C_m^i > qE^i X + S^i$. In this case, the income from crops and the money transfers enable the household to purchase game meat from other poachers.

The decision problem of the household is to decide the optimal hunting effort E_h^i and consumption of agricultural crops C_c^i in order to maximize its utility, subject to the

[4] Hunting performed by the local people is illegal. However, the probability of being caught in illegal hunting and the resulting costs, such as imposed fine, imprisonment etc., are ignored in this analysis. When analysing the cross-sectional data set, these components are fixed and equal for every household and, therefore, omitted.

constraints in *(3)*, *(4)* and *(5)*[5]. The first order conditions for maximum are given in Appendix 1. The resulting equation for the optimal hunting effort can be expressed as

$$(6) \qquad E_h^i = E(P_c, P_m, q, X, L^i, DC^i(X), S^i, M^i, Z^i) \geq 0$$

An interior solution for E_h^i (>0) emerges when effort is directed to hunting until the marginal benefit of hunting equals the marginal cost (see Appendix 1). A corner solution takes place, i.e. $E_h^i = 0$, if the marginal cost of hunting exceeds the marginal benefit. The latter will be the case if the wildlife-induced damage to crops is 'low' and the price of agricultural crops relative to the price of game meat is 'high'.

The household may produce agricultural crops for own consumption and trade any surplus on the market. However, the market fails for a particular household when it faces a 'low' price at which it can sell a crop. If this is the case, the household is better off by producing crops for own consumption only. That is, the constraint on crop consumption in *(4)* is binding. On the other hand, the individual household will sell excess crops on the market if the crop price is 'high'. Then the production of crops exceeds the household consumption and, consequently, the constraint on crop consumption is non-binding. Recall from above that a 'high' relative price P_c / P_m, all else equal, drives the hunting effort towards zero. A higher crop price may therefore encourage the household both to sell crops on the market and to reduce the effort use in illegal hunting.

Given the optimal hunting effort and household consumption of crops, the consumption of game meat follows from the budget constraint in equation *(5)*. The optimal effort in crop production results from the time constraint in equation *(3)*. The comparative static is derived in Appendix 1. The sign of the respective derivatives depends on whether the

[5] The wildlife stock X is treated as exogenous in this model. In the same way as Lopez (1998), it is assumed that the household does not account for the stock effect when deciding upon the optimal hunting effort. Moreover, the ecological dynamics are not of focus in the present analysis. Instead, we investigate the labour use in illegal hunting and interpret more labour use as an increased pressure on the wildlife stock. See also the empirical specification of *(6)* in section 4.1.

constraint on crop consumption is binding. Assume first that the constraint is non-binding, i.e. the household sells excess crops on the market. In this case, the household behaves as if the production and consumption decisions are made sequentially. That is, in the first step the household solves the production problem by maximizing the income from crop production and illegal hunting. This gives the budget constraint. In the second step the household determines the optimal consumption bundle subject to the budget constraint (see Sadoulet and de Janvry 1995).

The comparative static results are straightforward with a non-binding constraint on crop consumption. First, the hunting decision is independent of the amount of game meat distributed from the culling programme. This result stems from the fact that increased endowment of meat represents a lump-sum transfer when there are competitive markets for both agricultural crops and game meat. Second, the household reacts to an increased market price for crops by directing less effort towards hunting. In the same way, more agricultural land increases the marginal productivity of labour in crop production which leads the household to increase the effort use in this activity at the expense of hunting. In contrast, the household responds to increased wildlife-induced damage to crops by directing more effort towards hunting. This is also the effect of a higher price for meat. Finally, the time spent hunting increases as the endowment of time, i.e. number of household members able to work, increases.

The effects are somehow different when the constraint on crop consumption is binding, i.e. when the household utilizes harvested crops for own consumption only. Contrary to the non-binding scenario, the household directs less effort towards hunting when a larger amount of game meat is distributed from the culling programme. The mechanism works as follows. First, the household substitutes its consumption of illegal game meat with legal game meat. Second, the income effect works in the direction of an increase in domestic crop consumption. Both effects lead the household to devote less effort on hunting.

In contrast to the non-binding scenario, the hunting decision is now made independently of the market price for agricultural crops. Obviously, this result holds only for an upper limit where the household choose not to sell crops on the market. The effect of increased endowment of agricultural land on hunting effort is ambiguous. First, more cultivated land increases the output of crops for a given E_c^i. The household moderates the following effect on crop consumption by reducing its effort use in crop production. This is the direct effect and works in the direction of increased hunting effort. On the other hand, more land increases the marginal productivity of labour in crop production, which leads the household to devote less effort towards hunting. The total effect is therefore unclear. In the same way, increased damage in crop production has an ambiguous effect on hunting effort. The effect of a changing meat price is also ambiguous and strictly dependent on whether the household receive income transfers from tourism, i.e. whether it is a net consumer of meat. For net consumers ($Z^i > 0$) a higher meat price reduces the real budget available to buy meat. Consequently, the household increases the hunting effort to substitute purchased meat with own illegal offtake. In contrast, a higher price of game meat increases the income for net producers of meat ($Z^i < 0$). In this case, the household diverts less effort towards hunting. Finally, the effect of increased endowment of labour is positive and similar to the case of a non-binding constraint on crop consumption.

3. Data collection and descriptive analyses

3.1. Data collection

The empirical analysis of wildlife hunting is based on survey data from the Serengeti and Bunda Districts in Tanzania. Data was collected among 297 households in 6 villages of which half of the sample households live in villages which participate in the Serengeti Regional Conservation Project (SRCP). 166 households are from Bunda, while 131 households are from Serengeti. For a further description of the survey, see Appendix 2. The questionnaire deals with economic conditions and activities centred around the human-wildlife interface, such as wildlife hunting, household income, and wildlife-induced damage to crops and domestic animals, all for the year 2000. The households were also asked whether they participate in agricultural crop production for own consumption or as an income generating activity. The data on crop production cover mainly seven different crops: cotton, maize, sorghum, cassava, millet, potatoes and beans. Cotton is the only cash crop, while the food crops are produced both for sale on the market and as food for the household, or for household consumption only. Among the food crops, maize is the main income-generating crop. In addition to crop production and domestic animal keeping, 37 per cent of households in the study area earn income from selling fish, charcoal, local brew, running small shops etc. In the following, these activities will be referred to as 'other' activities[6]. Finally, to add to income and domestic consumption, people go hunting.

One main purpose of the empirical analysis is to investigate SRCP's impact on the illegal hunting activity in this area. This project currently includes fourteen villages spread evenly between Serengeti and Bunda Districts. The selection of the project villages has not been based on thorough studies of illegal activities, but is based on their closeness to the western borders of Serengeti National Park. As mentioned, SRCP's main strategy is to manage the game-culling programme. The culling quota is set as equal for each project village and determined by the government, i.e. the Ministry of natural resources and tourism[7]. The responsibility of SRCP is to organize the hunting and distribute the offtake to the respective villages. The villagers buy the meat at a price set in agreement between SRCP and the village authorities[8].

In addition to game meat distribution, SRCP has assisted the establishment of village-level institutions responsible for managing the fund from the hunting quota revenues. These funds finance village projects such as schools and dispensaries which, in turn, has reduced the individual tax burden. SRCP is also responsible for the set-up and training of game scouts in the project villages. Finally, SRCP works with awareness building in order to improve the relationship between the local people and the park. This includes public meetings at village level, seminars and training courses on wildlife utilization and management etc. For a broader overview of the activities of SRCP, see Rugumayo

[6] The complete list of 'other' activities also includes selling water, honey, and fruit, house rent, carpentry, making spears, and employment (teaching or other work at school, wildlife management, village secretary, other employment). Only 8 respondents in the sample households (less than 3 per cent) report that they have formal employment.

[7] For the year 2000 hunting season the quota was 15 wildebeest, 10 zebra and 5 topi for each project village.

[8] This price is set below the price of illegal game meat. The price of meat from SRCP was 400 tzh per kg in year 2000, while the mean reported price of illegal meat was some 600 tzh per kg (year 2001).

(1999)[9]. In the present analysis, four of the sample villages participate in SRCP[10], while there exists no village project in the sample villages outside SRCP[11].

3.2. Descriptive analysis and the sample

The households were asked about their participation in illegal hunting, hunting trips and travel distance to the hunting area. The data on hunted species covers wildebeest, zebra, gazelle, topi, and impala. Table 1 shows that 80 households, or 27 per cent of the sample, report that some household members are involved in illegal hunting. The participation rate in illegal hunting differs between sub-groups of the sample. For instance, the participation rate is 32 per cent among SRCP households and 22 per cent for households outside SRCP. These differences demonstrate the need for a further investigation of the impact of SRCP on illegal hunting. In addition, the participation rate varies between the districts, 22 per cent in Bunda District and 34 per cent in Serengeti District.

Table 1: Distribution of reported participation in hunting.

		Hunting	
	Number	Participation	No participation
Total sample	297	80 (27%)	217 (73%)
SRCP	148	47 (32%)	101 (68%)
Not SRCP	149	33 (22%)	116 (78%)
Bunda District	166	36 (22%)	130 (78%)
Serengeti District	131	44 (34%)	87 (66%)

Table 2 demonstrates that we can divide the hunters into two groups. This division is also important for the empirical specification of the model (see section 4.1). We have one group of hunters who report that they go on hunting trips and a second group of

9 SRCP intends to assist with loans and other kind of support to promote income-generating projects among the project villages. However, this is a small-scale project which is reflected by the fact that only 9% of the sample households participating in SRCP report that they benefit from this kind of assistance.

10 The project was implemented in Robanda and Nyakitono (Serengeti) in 1993, and in Nyamatoke and Mariwanda (Bunda) in 1994.

11 The village excecutive secretary in both Bukore and Rwamchanga village confirmed the absence of village projects.

hunters who don't go on hunting trips. 55 per cent of the hunters go on hunting trips, i.e. trips that last for several days and where the hunters usually hunt within the protected area. Wildebeest is the major target for this group, followed by zebra and gazelle. The remaining 45% of the hunters hunt closer to their homes and within the village area. For this group, hunting is carried out during the annual wildebeest migration when wildebeest enters village land during the dry season[12]. Several of these households report that they kill wildebeest when they enter their agricultural field or yard. This indicates that hunting in the home area is less time consuming than going on hunting trips.

Table 2: Distribution of the households involved in hunting.

	Total	Households with zero trips	Households with a positive number of trips
Number	80	36 (45%)	44 (55%)
SRCP	47	27 (57%)	20 (43%)
Not SRCP	33	9 (27%)	24 (73%)
Bunda District	36	5 (14%)	31 (86%)
Serengeti District	44	31 (70%)	13 (30%)

As seen in Table 2, the fraction of the hunters reporting a positive number of hunting trips differs between sub-groups of the sample. For instance, 43 per cent of the hunters in the SRCP villages report that they go on hunting trips, while the same rate for hunters outside SRCP is 73 per cent. The rates differ even more between the districts: 86 per cent of the hunters in Bunda go on hunting trips, while only 30 per cent of the hunters in Serengeti report the same. When it comes to the motivation for hunting, both groups of hunters report that they hunt both as a source of income and for domestic consumption. However, the groups differ when it comes to the reported income from illegal hunting. While 96 per cent of the households going on hunting trips earn income from this activity, this only applies to 33 per cent of those who hunt in their home area[13]. One

[12] See Sinclair and Arcese (1995) for a description of the wildebeest migration.
[13] It is important to note that distinguishing between hunting in the protected area and the village area as done here must not be confused with the terms 'subsistence' and 'organized' poaching used by Leader-Williams and Milner-Gulland (1993). The 'organized' poacher gangs of Leader-Williams and Milner-Gulland originate from

plausible explanation of the observed deviation in income is that the average offtake is considerably higher among households who go on hunting trips (see Table A1 in Appendix 3).

Between the districts, the data set reveals different participation rates in three income-generating activities, where some households earn income from several of these activities. While Table 3 clearly shows that crop production is the most common income generating activity among the sample households, the rate of households with income from this activity differs between the districts. All of the households in Bunda District possess agricultural land and 86 per cent earn income from crop production. 81 per cent of the Bunda households earning income from crops grow cotton, while 52 per cent grow maize.

Table 3 also shows that crop production is a major activity in Serengeti where 60 per cent of the households earn income from crops. In comparison with Bunda, only 11 per cent of the Serengeti households who earn income from crops devote land for cotton, while 94 per cent of these households grow maize. It turns out that the mean income from crops is significantly higher in Bunda than in Serengeti (see Appendix 3, Table A2), a result which may partly be due to the discovered difference in crop composition: while cotton is the only cash crop in the study area, maize is produced for both domestic consumption and as a source of income.

Table 3 shows that animal keeping is the second major activity in the study area. The households mainly hold cattle, goat, sheep, and poultry. The rate of households with a positive income from animal keeping is higher in Serengeti than in Bunda. Moreover, the mean income from animal keeping is significantly higher among the Serengeti households (see Appendix 3, Table A2). Finally, as seen in Table 3, 110 households earn income from other activities than crop production and domestic animal keeping.

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outside the local community of the study area Luangwa Valley, Zambia. In addition, they use more sophisticated hunting methods (i.e. automatic weapons) and hunt more often for trophies (i.e. elephant and rhino), than the subsistence hunters. Here, however, all hunters originate from the local community, they all hunt for meat (for domestic consumption or to sell), and they all use traditional hunting methods (i.e. wire snares, pitfall traps, knives, machetes etc. (see Arcese et al. 1995)). Therefore, in line with Leader-Williams and Milner-Gulland's terminology, both groups of hunters in this survey are subsistence hunters.

Again, the rate of participation differs between the districts, 40 per cent in Bunda and 33 per cent in Serengeti. The mean income from ‘other’ activities is significantly higher in Serengeti (see Appendix 3, Table A2). However, we find no significant difference in the mean total income (except hunting) between the districts. Hence, while the districts differ in type of income generating activities, there is no significant difference in the mean income level.

Table 3: Number of households earning income from the various activities.

Activity:	Crop	Domestic animals	‘Other’*
Total sample	220	153	110
Bunda District**	142 (86%)	74 (45%)	67 (40%)
SerengetiDistrict**	78 (60%)	79 (60%)	43 (33%)
SRCP**	100 (68%)	58 (39%)	66 (45%)
Not SRCP**	120 (81%)	95 (64%)	44 (30%)

*‘Other’ income does not include hunting.
**Per cent of the number of sample households in the respective sub-group.

When grouping the households by participation in SRCP, Table 3 shows that the rate of households with income from crops, domestic animals and other activities differ between the sub-groups. The rate of households earning income from crops and/or domestic animals is lower among SRCP households than those outside SRCP. Moreover, the mean income from each of these activities is significantly lower among the SRCP households (see Appendix 3, Table A2). In contrast, a higher rate of the SRCP households earns income from ‘other’ activities. However, the mean income from these activities does not differ significantly between the sub-samples. Still, the mean total income is significantly higher outside SRCP (see Appendix 3, Table A2).

The households were asked to indicate the level of wildlife-induced damage to crops and domestic animals as ‘no damage’, ‘very little’, ‘much’ or ‘very much’. Table 4 reports the answers. The second row shows that some 86 per cent of the respondents complain that wildlife causes ‘much’ or ‘very much’ damage to crops. This number seems high, and a further investigation of the percentage damage reported by the individual household shows a considerable variation within each response category. However, the survey reveals that the mean percentage damage increases between the categories and the means differ significantly. Still, there are some serious measurement problems regarding both of the reported measures of crop damage. First of all, the respondents may over-estimate the damage in the hope for future compensations. Second, the individual respondent estimated the percentage damage to his crops as the number of cultivated acres damaged relative to the number of acres cultivated. This may cause both over and under estimation of the money value of the damage as one acre of cotton (cash crop) is given the same weight as e.g. sorghum (food crop).

Table 4: Distribution of reported wildlife-induced damage to crops and domestic animals.

Response categories:		No damage	Very little	Much	Very much	Total	P*
Crop damage	Number of respondents	24	18	72	180	294	
	% of respondents	8.2	6.1	24.5	61.2	100	
	Mean % damage	1.7	12.3	17.8	22.6	19.1	0.000
Damage domestic animals	Number of respondents	73	12	70	55	210	
	% of respondents	34.8	5.7	33.3	26.2	100	
	Mean poultry lost/injured	1.2	2.7	5.5	9.4	5.1	
	Mean livestock lost/injured**	0.25	2.3	2.0	3.4	1.9	

*P is the observed significance level. The null hypothesis of equal means is rejected for $P \leq 0.05$
**Here, 'livestock' includes cattle, goats and sheep.

Table 4 also shows the distribution of the reported damage to domestic animals. As seen in the fifth row, some 60 per cent complain that wildlife causes 'much' or 'very much' damage. Compared to the reported crop damage, far more households respond that they experience 'no damage' to domestic animals. When it comes to the number of animals killed or injured by wildlife, we distinguish between damage to poultry on one hand and damage to bigger animals like cattle, goats and sheep on the other. The term livestock in Table 4 refers to bigger animals. The reported numbers vary considerably within each respond category. Some inconsistency may be present, but the variation may also reflect varying dependence on domestic animal keeping among the households.

4. Empirical specification and estimation results

4.1. Empirical specification

The sample used in the following empirical analysis is limited to the 80 households who report that they are involved in illegal hunting. In equation *(6)* the hunting effort E_h was defined as time spent on illegal hunting. The data set provides information on the number of hunting trips in 2000. In addition, for households reporting that they go on hunting trips, the data set states the average number of days per trip. However, for households hunting in their home area only (i.e. involved in illegal hunting but no trips, see Table 2), we lack information about how much time they spend hunting. For all these reasons, hunting effort is defined as the number of hunting trips, where $E_h > 0$ for those who go on hunting trips, while $E_h = 0$ for those who hunt in their home area. This means that the empirical analysis is related to the number of hunting trips rather than the actual time spent hunting. Still, this specification of the dependent variable is reasonable because the offtake is significantly higher among hunters who go on hunting trips (see Appendix 3, Table A1). Hence, there seems to be a potential for reduced aggregate illegal offtake when stimulating reduced hunting effort, i.e. reduced number of hunting trips per year.

Because we have data on the actual number of hunting trips for the households with a positive number of trips, we specify the empirical model as a Tobit model. The basic equation to be estimated is given in *(7)* where the number of hunting trips in household *i* $E_h^{i^*}$ is positive for the 44 sample households who go on hunting trips, while $E_h^{i^*} = 0$ for the 36 sample households who hunt within their home area[14].

[14] See Johnston and Dinardo (1997) chapter 13.

$$(7) \qquad E_h^{i^*} = \begin{cases} E_h^i & \text{if } E_h^i > 0 \\ 0 & \text{if } E_h^i = 0 \end{cases} \text{, where}$$

$$E_h^i = \beta_0 + \beta_1 SRCP + \beta_2 DISTRICT + \beta_3 L^i + \beta_4 Y^i + \beta_5 DC^i + \beta_6 DY^i + \beta_7 M^i + u^i$$

Note that the wildlife stock X is omitted from the empirical model. This is done because there exists no data on the distribution of the wildlife stock in relation to the location of the sample villages. Consequently, we must consider the wildlife stock as equally distributed within the study area and, hence, the stock size is equal for every village and household and, therefore, omitted from the model[15].

The explanatory variable *SRCP* in *(7)* is specified as a dummy for participation in the SRCP project. *DISTRICT* is a dummy for the district and is included in order to capture district-specific characteristics of the data set. The districts differ in several ways. First, and as already discussed, the crop composition in agricultural production varies in that cotton is produced mainly in Bunda. Second, the size of the human population in the sample villages in Bunda exceeds the one of the Serengeti villages, which may imply a bigger local market in Bunda. In addition, the villages in Bunda are located closer to a city, i.e. Bunda Town, and the market place there. The sample villages in Serengeti are located closer to the park management authority's headquarters, which may influence the interaction between the management authority and the local people. Because of these differences, it is adequate to control for the district in the empirical analysis.

The explanatory variable L^i is the number of acres cultivated for crop production in household *i*. DC^i is a discrete variable on crop damage reflecting the four response categories ranging from 'no damage' to 'very much damage'. As seen in Table 3, domestic animal keeping is a common activity among the sample households. Therefore, we introduce the size of the herd Y^i, measured by the number of domestic

[15] See Campbell and Borner (1995) for estimates of the wildlife population in the Serengeti ecosystem.

animals, and wildlife-induced damage to this herd DY^i as explanatory variables. DY^i is a discrete variable in the same way as for the crop damage. We expect that an increased size of the domestic stock and/or reduced damage to this stock will increase the marginal return from animal keeping and stimulate the household to direct less effort towards illegal hunting. M^i is the number of household members and we expect that the hunting effort increases with the size of the household. However, it is important to note that this explanatory variable counts all members of the households, frequently ranging from small children to elders. Consequently, this is not an adequate measure of number of household members capable of working. Still, it is worthwhile to investigate the impact of this kind of household characteristic. Finally, u^i is the error term. Summary statistics of the variables are reported in Table A2 in Appendix 3.

Equation *(7)* is the basic empirical model. However, later this model will be modified in order to capture patterns in crop production and domestic animal keeping, income from other legal activities than agricultural production, and game meat bought from SRCP. When investigating the impact of domestic animal keeping we will distinguish between the poultry and bigger animals like cattle, goat, and sheep. $POULTRY^i$ counts the number of poultry in household *i*, while the variable $LIVESTOCK^i$ measures the number of domestic animals except poultry. The explanatory variable $OTHER^i$ captures the income from other activities than agricultural production and illegal hunting in household *i*. We will also introduce the variable $MEAT^i$ which indicates the number of kilo game meat bought from SRCP in household *i* year 2000. This is a discrete variable ranging from '<5kg', '[5, 10]kg', '[11, 20]kg', '[21, 30]kg', to '>30kg'.

Explanatory variables for the number of acres directed to the production of cotton L^i_COT and maize L^i_MAI will also be included. In addition, because some crops are produced for both own consumption and the market, it is necessary to compute an index working as a proxy for the degree of crop production for own consumption in the respective household. The households were asked to indicate whether a particular crop is produced for own consumption and/or for sale on the market. They also specified how many acres of land they devote for each crop. The index is based on this information and defined as follows. First, let $L_k^{j,i}$ be the number of acres devoted to crop $j \in [1, v]$ by household $i \in [1, n_k]$ in district $k \in [0,1]$. Then, the total number of acres cultivated for crop production in household *i* in district *k* is $\sum_{j=1}^{v} L_k^{j,i}$. Second, let ω_k^j be the share of the households in district *k* who produce crop *j* for own consumption only. The index of crop production for own consumption in household *i* district *k* is then given by

$$(8) \qquad INDEX_k^i = \sum_{j=1}^{v} \omega_k^j L_k^{j,i} / (\sum_{j=1}^{v} L_k^{j,i})$$

The index is district specific in the sense that it is based on consumption shares in the respective district. An increase in $INDEX_k^i$ means that a higher share of crop production in household *i* in district *k* is used for own consumption only.

4.2. Estimation results

Table 5 reports the Tobit estimates for the basic model in *(7)* as well as the additional regressions. The coefficient of the political variable *SRCP* in the basic regression (a) has a significant negative sign, which suggests that SRCP has succeeded in reducing the number of hunting trips in the study area. This is also the case for regressions (c) to (d). However, we cannot state from which activities of SRCP this result stems from. For instance, the dummy variable *SRCP* may reflect the culling programme where game meat is distributed to the project villages. In order to investigate the impact of game meat distribution, the SRCP households who report that they buy game meat from this programme (i.e. 94 per cent) were asked to estimate the number of kilo meat bought (see section 4.1). This gives the discrete variable *MEAT* in regression (e). The coefficient is not significantly different from zero, which means that we cannot reject the null hypothesis saying that game meat distribution has no effect on hunting effort. Instead, the significant negative sign of *SRCP* in regression (a)-(d) may reflect the presence of village game scouts and the establishment of village wildlife funds which may have reduced the antagonism towards wildlife in the SRCP villages. It is also possible that SRCP's attempts on awareness building have affected the villagers' attitude towards wildlife in the same direction.

The coefficient of *L* in the basic regression (a), the amount of land cultivated for crop production by the household, is negative but not significantly different from zero. As discussed in section 3.1 the crop composition differs between the districts in that cotton – the main crop in Bunda – is produced for the market, while maize – the main crop in Serengeti – is produced both for the market and household consumption. Regression (b) controls for the different patterns of crop production by distinguishing between land devoted to cotton and land devoted to maize. The coefficient of *L_COT* comes out as negative and significantly different from zero. The coefficient of *L_MAI* is also negative, but only significant on the 10 per cent significance level. Both results are in accordance with the theoretical model of a net producer of agricultural crops. The sign of the coefficient of *L* in regression (b) is positive but not significant. However, the sign differs from regression (a), which may indicate that the hunting effort increases if additional land is used for a subsistence food crop. Therefore, regression (c) makes a distinction between crop production for own consumption and crop production for the market by introducing *INDEX* as an independent variable. The coefficient is positive and significantly different from zero, which means that peasants with a higher degree of subsistence crop production devote more effort towards hunting. In order to reduce the hunting effort, the policy implication is therefore to provide for easier access to the market for agricultural crops.

The underlying theoretical model predicts that wildlife-induced damage to crops leads to an increase in the hunting effort when the peasant is a net producer of crops, whereas the effect is ambiguous when crops are produced for own consumption only. The estimated coefficients in regressions (a) to (d) square with the hypothesis that damage to crops increases the hunting effort. The sample in regression (f) counts the households reporting that they earn no income from crop production, i.e. crop production for own consumption only. Also here the coefficient comes out as positive and significantly different from zero.

Regressions (a) to (c) report that the impact on the hunting effort of more extensive wildlife-induced damage to domestic animals is positive and significantly different from zero. This result, together with the findings for crop damage above, supports the widespread view that damage induces a shift in labour use towards wildlife extraction (see Skonhoft and Solstad (1998) for a theoretical analysis and Kiss (1990) for a broader discussion of the costs of living with wildlife). This suggests that policies which support a tighter damage control, such as fencing, chasing problem animals out of the villages and so forth, will reduce the hunting pressure and increase the degree of wildlife conservation.

In regressions (a) to (c) the number of domestic animals comes out with a negative sign and the coefficient is significantly different from zero. There are two possible reasons for this result: first, more domestic animals may increase the time spent herding and, second, it may reduce the consumption of game meat via a substitution effect, both of which lead the household to devote less effort towards hunting. Regression (d) distinguishes between livestock (i.e. cattle, goat, and sheep) and poultry keeping and demonstrates negative significant effects of both. Livestock herding is a relatively time-consuming activity and, consequently, the negative coefficient may reflect that the household responds to more livestock by increasing the labour use in this activity. In contrast, because poultry keeping is less time-consuming, the negative coefficient may reflect that poultry is a substitute for game meat in consumption.

The theoretical model assumes that the household is involved in two activities only; crop production and illegal hunting. Above, we also looked at the estimated effects of domestic animal keeping. However, 46 per cent of the sample households report that they earn income from other activities as well. In regression (b) to (d) the coefficient of the income from these activities *OTHER* is positive, but not significantly different from zero, which supports the theoretical model.

Table 5: Estimation results. Tobit model. Dependent variable E_h.
t-values in parentheses
SRCP: 0 = no, 1 = yes, DISTRICT: 0 = Serengeti, 1 = Bunda

	(a)	(b)	(c)	(d)	(e) SRCP only	(f) Crop income =0
SRCP	-4.195	-4.414	-4.197	-4.211		-4.513
	(-4.63)	(-5.13)	(-5.07)	(-5.09)		(-2.20)
MEAT					0.888	
					(0.70)	
DISTRICT	3.914	4.185	3.922	3.945	4.413	6.560
	(4.63)	(4.80)	(5.29)	(5.33)	(3.47)	(2.72)
L	-0.125	0.259			0.418	0.313
	(-0.76)	(1.37)			(1.06)	(0.82)
L_COT		-0.791				
		(-2.19)				
L_MAI		-0.551				
		(-1.76)				
INDEX			6.379	6.007		
			(2.49)	(2.26)		
Y	-0.156	-0.165	-0.159		-0.193	-0.232
	(-3.53)	(-3.98)	(-4.21)		(-2.14)	(-2.03)
POULTRY				-0.172		
				(-3.77)		
LIVESTOCK				-0.144		
				(-3.04)		
DC	1.918	2.170	1.825	1.820	5.980	6.666
	(2.77)	(3.34)	(2.94)	(2.97)	(3.02)	(2.45)
DY	1.790	1.665	1.904	1.905	2.598	3.250
	(4.73)	(4.42)	(5.42)	(5.44)	(3.62)	(3.26)
OTHER		-0.029	-0.021	-0.023		
		(-1.07)	(-0.80)	(-0.85)		
M	-0.297	-0.316	-0.313	-0.317	-1.108	-0.400
	(-2.01)	(-2.37)	(-2.64)	(-2.68)	(-2.83)	(-0.71)
# obs.	58	57	55	55	34	25
R^2_{adj}	0.270	0.314	0.328	0.329	0.286	0.386

Note: a coefficient is significantly different from zero at level of 5 per cent for $|t| > 2$.

5. Discussion and concluding remarks

The incentives to hunt illegally are detrimental for wildlife conservation in protected areas in developing countries. Despite of this, little empirical attention has been paid to the underlying motivation for illegal hunting. Knowledge about this issue is crucial for providing sound advice to policymakers in order to reach the joint aim of wildlife conservation and economic development.

This paper estimates functions for labour effort in illegal hunting, where the number of hunting trips serves as a proxy for labour use. The analysis provides several policy recommendations that have the potential of reducing the number of hunting trips and thereby promoting wildlife conservation. Cross-sectional data from a household survey in western Serengeti is used to identify factors determining the labour use in illegal hunting in this area. First, the empirical results suggest that the establishment of the ICDP in western Serengeti, i.e. the Serengeti Regional Conservation Project (SRCP), has reduced the illegal hunting pressure. However, it is important to note that we cannot draw a conclusion on the conservation effect of SRCP based on this result alone. SRCP exercises game culling and we do not know how this has affected the aggregate wildlife offtake. Further investigations of the extent of the culling programme relative to the illegal offtake is therefore of major importance.

Second, the empirical analysis reveals another important relationship, namely that hunting in western Serengeti is related to the patterns in agricultural crop production. The estimation results demonstrate that a higher degree of cotton production and other crop production for the market stimulate to increased effort in agricultural production and a reduced illegal hunting pressure. In Bunda District, policies which stimulate increased cotton production, e.g. more extensive use of pesticides and irrigation systems, have the potential of reducing the hunting pressure.

The number of cotton buyers visiting Bunda District during the harvest period reflects that the access to the market for cotton is relatively easy for the households in this district. However, the situation is somehow different in Serengeti District, where people complain about poor access to market. It is therefore important to stimulate increased accessibility to market for the households in this district. One important challenge is to improve the infrastructure in the area, for instance through road construction, in order to reduce the transaction costs. In addition, support of fertilizer use and pesticides will reduce the costs of crop production and, hence, encourage the peasants to participate in the market. However, policymakers should be aware of possible trade-offs between such improvements and environmental degradation. For instance, road construction may facilitate the illegal transport of wildlife products out of this area and make them more tradable. Infrastructure improvements should therefore be combined with more extensive use of anti-poaching law enforcement.

Wildlife imposes damage to crops and domestic animals and the empirical results indicate that this increases the illegal hunting. This result should encourage policymakers to make initiatives to reduce and prevent wildlife-induced damage, such as support of fencing, chasing problem animals out of the villages, and so forth. Another option is to compensate the local peasants for the costs of living with wildlife. There are, however, some obvious pitfalls related to this strategy; people may overestimate the damage and a compensation scheme may attract people from other areas and thereby increase the human pressure on the park borders.

We have also seen in this analysis that the hunting pressure is negatively related to domestic animal keeping. Policies which stimulate the local people to keep more animals will therefore reduce the hunting pressure in the study area. However, policymakers should be aware that more livestock (i.e. cattle, goat, and sheep) means more intensive grazing, which may reduce the quality of the wildlife habitat. The long-term consequence for wildlife conservation is therefore highly unclear. However, poultry does not compete with wildlife in the same way as grazing animals. A better strategy for promoting wildlife conservation is therefore to support poultry keeping.

In summary, the empirical results suggest that SRCP has succeeded in reducing the illegal hunting pressure in Serengeti. Other initiatives that may promote wildlife conservation include attempts which encourage a higher degree of crop production for the market and more extensive use of damage control. Moreover, the empirical results suggest the best strategy for achieving a joint objective of wildlife conservation and economic development. While the implementation of SRCP has reduced the illegal hunting pressure, we do not know the impact on the wildlife stock. In addition, records from SRCP show an average expected revenue from the culling programme of 834 000 tzh per village in 2000, or some 2 300 tzh per household. This is low compared to the potential return from agriculture where the average income from crops among the cotton producers was 88 000 tzh (see Appendix 3, Table A4). These numbers indicate that the individual income-advantage of participating in SRCP is highly limited. In order to promote both wildlife conservation and local welfare policymakers should instead make arrangements which encourage and ease the access to markets for agricultural crops.

References

Arcese, P. et al. (1995): "Historical and present-day antipoaching in Serengeti". In Sinclair, A.R.E. and P. Arcese (eds): *Serengeti II*, University of Chicago Press, Chicago, 506-533.

Barrett, C.B. and Arcese, P. (1995): "Are integrated conservation-development projects (ICDPs) sustainable? On the conservation of large mammals in Sub-Saharan Africa", *World Development*, 23(7), 1073-84.

Barrett, C.B. and P. Arcese (1998): "Wildlife harvest in integrated conservation and development projects: linking harvest to household demand, agricultural production, and environmental shocks in the Serengeti", *Land Economics*, 74(4), 449-65.

Brandon, K.E. and M. Wells (1992): "Planning for people and parks: design dilemmas", *World Development*, 20(4), 557-570.

Brown, K. et al. (1993): "Economics and the conservation of global biological diversity", *Global Environmental Facility, Working Paper*. No. 2, The World Bank, New York.

Bulte, E.H. and D. van Soest (1999): "A note on soil depth, failing markets and agricultural pricing", *Journal of Development Economics*, 58(1), 245-254.

Campbell, and Borner (1995): "Population trends and distribution of Serengeti herbivores: implications for management". In Sinclair, A.R.E. and P. Arcese (eds): *Serengeti II*, University of Chicago Press, Chicago, 117-145.

Campbell, K. and H. Hofer (1995): "People and wildlife: Spatial dynamics and zones of interaction". In Sinclair, A.R.E. and P. Arcese (eds): *Serengeti II*, University of Chicago Press, Chicago, 534-570.

Carlson, G.A. and M.E. Wetzstein (1994): "Pesticides and pest management". In Carlson, G.A., D. Zilbermann, and J.A. Miranowski (eds): *Agricultural and Environmental Resource Economics*, Oxford University Press, Oxford, 268-318.

De Janvry, A., M. Fafchamps, and E. Sadoulet (1991): "Peasant household behaviour with missing markets: Some paradoxes explained", *Economic Journal*, 101(409), 1400-1417.

Gibson, C.C. and S.A. Marks (1995): "Transforming rural hunters into conservationists: An assessment of community-based wildlife management programs in Africa", *World Development*, 23(6), 941-957.

Johnston, J. and J. Dinardo (1997): *Econometric Methods*, McGraw Hill, Singapore.

Kiss, A. (1990): "Living with wildlife: Wildlife resources management with local participation in Africa." *Technical Paper*, 130, World Bank, Washington D.C.

Leader-Williams, N. and E.J. Milner-Gulland (1993): "Policies for the enforcement of wildlife laws: The balance between detection and penalties in Luangwa valley, Zambia", *Conservation Biology*, 7(3), 611-617.

Lopez. R. (1998): "Agricultural intensification, common property resources and the farm-household", *Environmental and Resource Economics*, 11(3-4), 443-458.

Rugumayo, C.R. (1999): "The participation in sustainable resource management. The case of Serengeti regional conservation strategy, Tanzania", Working Paper 112, Norwegian Institute for Urban and Regional Research, Oslo.

Sadoulet, E. and A. de Janvry (1995): *Quantitative Development Policy Analysis*, Johns Hopkins University Press, Maryland.

Sinclair, A.R.E. (1995): "Serengeti past and present". In Sinclair, A.R.E. and P. Arcese (eds): *Serengeti II*, University of Chicago Press, Chicago, 3-30.

Sinclaire, A.R.E. and P. Arcese (1995): *Serengeti II*, University of Chicago Press, Chicago.

Skonhoft, A. and J.T. Solstad (1998): "The political economy of wildlife exploitation", *Land Economics*, 74(1), 16-31.

SRCS (1993): "Implementation report of the development activities for the year 1993". Prepared for the Ministry of Natural Resources and Tourism, Wildlife Division, Dar es Salaam, Tanzania.

SRCS (1995): "Implementation report of the development activities for the financial year 1994/95". Prepared for the Ministry of Natural Resources and Tourism, Wildlife Division, Dar es Salaam, Tanzania.

Zivin, J., B.M. Hueth, and D. Zilberman (2000): "Managing a multiple-use resource: The case of feral pig management in California rangeland", *Journal of Environmental Economics and Management*, 39(2), 189-204.

Appendix 1: Optimisation and comparative static

Solving equation *(5)* for C_m^i, equation *(3)* for E_c^i and inserting in *(1)* yields

$$U^i = U\left\{C_c^i, \left[f(M^i - E_h^i; L^i)(1 - DC^i(X)) - C_c^i\right](P_c / P_m) + qE_h^i X + S^i + Z^i / P_m\right\}$$

The decision problem of the household is therefore to determine the optimal hunting effort E_h^i and crop consumption C_c^i in order to maximize its utility. The Lagrange function reads

$$V^i = U\left\{C_c^i, \left[f(M^i - E_h^i; L^i)(1 - DC^i(X)) - C_c^i\right](P_c / P_m) + qE_h^i X + S^i + Z^i / P_m\right\}$$
$-\lambda\left[C_c^i - f(M^i - E_h^i; L^i)(1 - DC^i(X)\right]$. The first order conditions are given by

(A1) $\quad U_m\left[qX - (P_c / P_m) f_1(M^i - E_h^i; L^i)\left[1 - DC^i(X)\right]\right]$

$\quad\quad -\lambda f_1(M^i - E_h^i; L^i)\left[1 - DC^i(X)\right] \leq 0$; $= 0$ if $E_h^i > 0$

(A2) $\quad U_c - U_m(P_c / P_m) - \lambda \leq 0$; $= 0$ if $C_c^i > 0$

where U_c and U_m denote the first order derivative with respect to C_c^i and C_m^i, respectively, while f_1 denotes the first order derivative with respect to E_c^i. $\lambda \geq 0$ is the shadow value of crop consumption and states that the utility in optimum either increases or is not affected by a positive shift in the net crop output, e.g. reduced crop damage. In the case of a non-binding constraint on crop consumption, the household sells excess crops on the market, which drives the shadow value to zero, $\lambda = 0$. When the market for crops fails, the consumption of crops is constrained by the net production in the sense that the household produces crops for own consumption only. This gives a positive shadow value, $\lambda > 0$. *(A1)* and *(A2)* determine E_h^i and C_c^i, while the effort use in agriculture E_c^i and the consumption of game meat C_m^i follow from equations *(3)* and *(5)*, respectively.

First, consider the case of a non-binding constraint on crop consumption. Then, in case of an interior solution, *(A1)* is reduced to

(A1') $qX - (P_c / P_m) f_1(E_c^i; L^i)\left[1 - DC^i(X)\right] = 0$

which determines the optimal hunting effort E_h^i. Differentiation of *(A1')* gives

$$\frac{dE_h^i}{dS^i} = 0$$

$$\frac{dE_h^i}{dP_m} = \frac{-f_1(E_c^i; L^i)/P_m}{f_{11}} > 0$$

$$\frac{dE_h^i}{dP_c} = \frac{f_1(E_c^i; L^i)}{P_c f_{11}} < 0$$

$$\frac{dE_h^i}{dL^i} = \frac{f_{12}(E_c^i; L^i)}{f_{11}} < 0$$

$$\frac{dE_h^i}{dDC^i} = \frac{-f_1(E_c^i; L^i)}{f_{11}} > 0$$

$$\frac{dE_h^i}{dM^i} = 1$$

Second, consider the case of a binding constraint on crop consumption, i.e. $C_c^i = f(E_c^i; L^i)(1 - DC^i(X))$ and $\lambda > 0$. Then *(5)* gives $C_m^i = qE_h^i X + S + Z^i / P_m$. Solving *(A2)* for λ and inserting this in *(A1)* gives the first order condition for an interior solution as

(A3) $U_m qX - U_c f_1(E_c^i; L^i)(1 - DC^i(X)) = 0$

Differentiation of *(A3)*, when accounting for the time constraint in *(3)*, gives the comparative static results ($\sigma < 0$ from the second order maximum condition for E_h^i):

$$\frac{dE_h^i}{dS^i} = \frac{-U_{mm}qX + U_{cm}f_1(1-DC^i)}{\sigma} < 0$$

$$\frac{dE_h^i}{dP_m} = \frac{\left[U_{mm}qX - U_{cm}f_1(1-DC^i)\right]Z^i / P_m^2}{\sigma}$$

$$\frac{dE_h^i}{dP_c} = 0$$

$$\frac{dE_h^i}{dL^i} = \frac{(1-DC^i)\left[U_c f_{12} + U_{cc}f_1 f_2(1-DC^i) - U_{mc}qXf_2\right]}{\sigma}$$

$$\frac{dE_h^i}{dDC^i} = \frac{U_{mc}qX - U_c f_1 - U_{cc}f(E_c^i; L^i)f_1(1-DC^i)}{\sigma}$$

$$\frac{dE_h^i}{dM^i} = \frac{(1-DC^i)\left[U_c f_{11} + U_{cc}f_1^2(1-DC^i) - U_{mc}qXf_1\right]}{\sigma} > 0$$

Appendix 2: The survey

During the period of June-August 2001 I conducted interviews in 297 households in Serengeti and Bunda Districts. In order to capture the human-wildlife interface, six villages located along the western border of the Serengeti National Park were selected for participation in the survey. The villages are Bukore, Mariwanda, and Nyamatoke in Bunda District and Nyakitono, Robanda, and Rwamchanga in Serengeti District. Four of these villages participate in SRCP, namely Nyamatoke, Mariwanda, Nyakitono, and Robanda. The households were picked at random from lists of names, and the number of households from each village was decided by weighting the villages by their respective size. In each household, whenever possible, the head of the household was interviewed. The interviews were conducted in Kiswahili with translation assistance from two local Tanzanians.

Based on experience from test interviews in Bukore, a strategy was developed on how to approach the questionnaire in general and especially the sensitive questions on illegal hunting. In order to gain confidence from the local people, we spent much time in the villages and had two inhabitants in each village to visit the households in advance and explain the purpose of the survey. The interviews took place in the home of the respective household.

The households were asked whether any of the household members were involved in hunting in 2000. Those who answered yes were asked additional questions in order to capture the extent of the hunting activity. None of the respondents have a licence to hunt, which means that all hunting recorded in this survey is illegal. People submitted information about the number of hunting trips, the number of days spent per hunting trip, and the average travel distance per trip. However, we discovered that some of the hunters hunt within their home area only, especially during the annual wildebeest migration. The questions on trips and distance were therefore omitted for this group.

Some caveats should be made as the data set have a few weaknesses that are common for questionnaires. First, information on income is likely to be understated because some respondents are suspicious and fear that the information will be handed over to the district and central government for taxation purposes. Second, the quantitative data on plot size under various agricultural uses are given by the respondent's subjective estimate, which may be subject to errors. The same applies to the estimated wildlife-induced damage to crops and livestock. The reader should be aware of these problems when reading the paper.

Appendix 3: Tables

Table A1: Mean wildlife offtake per household in 2000 for households involved in hunting. Kruskal-Wallis test.

	Hunting trips?		
	Yes	No	P*
Mean (st. dev.)	13.86 (30.39)	2.25 (1.99)	0.000
N**	N=43	N=36	

*P is the observed significance level. The null hypothesis of equal means is rejected for $P \leq 0.05$

**The third row reports the number of observations in the respective sub-sample. Here, one observation is missing among those who go on hunting trips.

Table A2: Mean income among households involved in the respective categories (1000 tzh). Standard deviations are reported in parenthesis. N is the number of observations in the respective sub-samples. Kruskal-Wallis test.

	Crop[1]	Domestic animals[2]	'Other'[3]	Total[4]
Bunda	87.85 (122.81) N=166	26.72 (50.24) N=122	73.91 (129.09) N=67	140.88 (171.23) N=166
Serengeti	44.72 (99.37) N*=129	43.39 (84.99) N=109	175.19 (261.40) N=43	137.80 (205.15) N=131
P[5]	0.000	0.005	0.028	0.203
SRCP	60.39 (96.79) N*=147	26.05 (43.48) N=113	73.45 (130.63) N=66	113.24 (135.02) N=148
Not SRCP	77.53 (130.36.) N*=148	42.76 (86.42) N=118	173.57 (258.11) N=44	165.62 (224.00) N=149
P[5]	0.009	0.017	0.210	0.048
Total	68.99 (114.99) N*=295	34.59 (69.21) N=231	113.50 (197.19) N=110	139.52 (186.63) N=297

1 Here N is the number of respondents in the respective sub-samples reporting that they cultivate land for crop production.
2 Here N is the number of respondents in the respective sub-samples reporting that they keep domestic animals.
3 Here N is the number of respondents in the respective sub-samples reporting positive income from other activities than crop production, animal keeping and hunting.
4 Here N is the number of respondents in the respective sub-samples
5 P is the observed significance level. The null hypothesis of equal means is rejected for $P \leq 0.05$.

Table A3: Data description and descriptive statistics.

Variable	Description	N	Mean (st. dev.)
MEAT	Kilo game meat bought from SRCP	45	2.20 (0.690)
L	Acres of land cultivated for crop production in the household	80	6.147 (5.236)
L_COT	Number of acres cultivated for cotton in the household	75	0.7400 (1.167)
L_MAI	Number of acres cultivated for maize in the household	75	1.437 (1.807)
INDEX	Index for the degree of crop production for own consumption [0,1]	72	0.702 (0.168)
Y	Number of animals in the household	75	13.60 (14.92)
POULTRY	Number of poultry in the household	75	7.63 (8.69)
LIVESTOCK	Number of livestock in the household	75	5.97 (12.52)
DC	Crop damage indicated from 1 (no damage) to 4 (very much damage)	80	3.50 (0.78)
DY	Poultry/livestock damage indicated from 1 (no damage) to 4 (very much damage)	63	2.37 (1.17)
OTHER	Income from non-agricultural activities (except hunting) year 2000, 1000 tzh	80	42.51 (112.13)
M	Household size, number of household members	80	6.09 (3.30)

Table A4: Mean income in 2000 (1000 tzh).
Standard deviation in parentheses. Kruskal-Wallis test.

	Cotton (N=129)	No cotton (N=142)	P*
Crop income	88.37 (85.94)	37.50 (92.56)	0.000
Total income**	147.43 (151.01)	125.37 (197.22)	0.001

*P is the observed significance level. The null hypothesis of equal means is rejected for $P \leq 0.05$

**Total income except income from hunting.

Table A5: Correlation matrix for variables in the equation for hunting effort

	Dep vari	SRCP	Meat SRCP	Dist-rict	Acre crops	Acre cotton	Acre maize	Index	No. anim	No. poult	No. livest	Crop damage	Domestic animal damage	Other inc	House memb
Dep vari	1.000														
SRCP	-.453	1.000													
Meat SRCP	-.033		1.000												
District	.611	-.309	.154	1.000											
Acre crops	-.124	-.159	.274	-.070	1.000										
Acre cotton	.215	-.187	.182	.460	.321	1.000									
Acre maize	-.198	-.433	.270	-.200	.363	-.194	1.000								
Index	-.035	.554	-.288	-.169	-.059	-.438	-.470	1.000							
No. animals	-.238	-.084	.210	-.175	.545	.002	.204	-.074	1.000						
No. poultry	-.080	-.261	.245	.017	.099	.094	.381	-.413	.456	1.000					
No. livestock	-.210	-.082	.110	-.205	.541	-.061	-.031	.195	.801	-.167	1.000				
Crop damage	.026	.274	.097	-.177	.128	.063	-.068	.013	-.032	.123	-.118	1.000			
Domestic animal damage	.502	-.264	.016	.227	.242	.266	-.105	-.150	.245	.218	.125	.027	1.000		
Other income	-.194	.105	.070	-.160	-.038	-.104	.026	.030	.071	.138	-.014	.053	-.267	1.000	
House memb	.033	-.100	-.015	.122	.413	.314	-.134	.068	.072	-.094	.143	-.021	.204	.071	1.000

Questionnaire

Village:

1 .... ☐
2 .... ☐
3 .... ☐
4 .... ☐
5 .... ☐
6 .... ☐

SRCP village?

Yes .... ☐
No .... ☐

SRCP village since year

1.a You are:

Man .... ☐
Woman.................. ☐

1.b Your age is:

1.c Your education is:

Never been to school................ ☐
Primary school .... ☐
Secondary school .... ☐
College/University .... ☐

1.d Number of household members:

2. Tribe

3.a Were you borne in this village?

Yes .... ☐
No .... ☐

3.b If no, why did you move to this village?

Family .... ☐
Land .... ☐
Employment................................ ☐
Game meat .... ☐
Firewood .... ☐
Other .... ☐ Explain ⇓

4.What income generating activities is your household involved in? Indicate, the approximate annual income from these activities.
(In thousand tzs.)

	Previous year (2000)
Agricultural crop production	000 tzs
Livestock/poultry keeping	000 tzs
Firewood	000 tzs

	Previous year (2000)
Other tree production	000 tzs
Handicraft	000 tzs
Industry	000 tzs
Employment in tourist lodges/camps, as guides etc	000 tzs
Employment in village game scouts	000 tzs
Other:	000 tzs

5.a How many acres of land does your household cultivate for crop production?

Before SRCP: [| | ,] Acres Year 2000: [| | ,] Acres

5.b How many acres do you cultivate for each crop?

Crop	Before SRCP	Year 2000
	[| | ,] Acres	[| | ,] Acres
	[| | ,] Acres	[| | ,] Acres
	[| | ,] Acres	[| | ,] Acres
	[| | ,] Acres	[| | ,] Acres
	[| | ,] Acres	[| | ,] Acres
	[| | ,] Acres	[| | ,] Acres

6. Which of these crops are produced for own consumption and which are produced for the market?

	Before SRCP		Today	
	Own	Market	Own	Market
Sorghum	☐	☐	☐	☐
Cassava	☐	☐	☐	☐
Maize	☐	☐	☐	☐
Cotton	☐	☐	☐	☐
Finger Millet	☐	☐	☐	☐
Potatoes	☐	☐	☐	☐
Beans	☐	☐	☐	☐
Rice	☐	☐	☐	☐
Tomatoes	☐	☐	☐	☐
	☐	☐	☐	☐
	☐	☐		☐

7. How many animals does your household have?

	Before SRCP	Year 2000	Today
Cattle			
Goats			
Chicken			
Pigs			
Sheep			
Other:			

8. Do you keep livestock/poultry for own consumption (including draught power), and/or to sell at the market?

	Before SRCP	Today
Own consumption	☐	☐
Market	☐	☐

9.a How much damage did wildlife cause to your crops during a year?

	Before SRCP	Year 2000
No damages	☐	☐
Very little	☐	☐
Much	☐	☐
Very much	☐	☐

9.b Please indicate, for the previous year (2000), this damage as a percentage of total crop production:

[| |] % of annual crop production

9.c How much damage/injury/loss did wildlife cause to your livestock/poultry during a year?

	Before SRCP	Year 2000
No damages	☐	☐
Very little	☐	☐
Much	☐	☐
Very much	☐	☐

9.d How many livestock/poultry were lost/injured by wildlife during the last year (2000)?

Livestock: ☐☐ Poultry: ☐☐

9.e How many poultry/livestock were lost/injured by birds/eagles last year (2000)?

Livestock: ☐☐ Poultry: ☐☐

9.f Is your livestock more often afflicted by illness in the period when wildebeest and zebra migrate through your area?

Yes .... ☐ No..................................... ☐

10.a Is any member of this household involved in hunting?

Yes .... ☐ No..................................... ☐

10.b If yes, what is your household's motivation for hunting?
Mark the alternatives relevant for your household.

	Before SRCP	Year 2000
Meat for the household	☐	☐
Meat for the market	☐	☐
Skin/trophies for own use	☐	☐
Skin/trophies for the market	☐	☐
Get rid of animals causing livestock and crop losses	☐	☐
Culture and tradition	☐	☐
	☐	☐
	☐	☐

15.a Does your household buy wildlife meat from SRCP?

Yes .... ☐
No .... ☐

15.b If yes, how many kilograms of meat did you buy from SRCP last year (2000)?

<5 .... ☐
5-10 .... ☐
11-20 .... ☐
21-30 .... ☐
>30 .... ☐

15.c Would you buy more if the price was lower?

Yes .... ☐ No...................................... ☐

15.d Do you buy game meat from other sources than SRCP?

Yes .... ☐ No...................................... ☐

If yes, specify:

16. How much does your household benefit from SRCP?

	No	Little	Much	Very much
Money transfers to your household	☐	☐	☐	☐
Reduced taxes and school fees for your household	☐	☐	☐	☐
Loan/help with handicraft etc	☐	☐	☐	☐
Village gets money for schools, hospital	☐	☐	☐	☐

Other:

17. Does your household have a hunting licence?

Yes .... ☐
No .... ☐

18. Did your household hunt more animals than legally permitted last year (2000)?

Yes .... ☐
No .... ☐

19. Has any member of your household been arrested because of hunting during a hunting trip during this or last year (2000-2001)?

Yes .... ☐
No............. ☐

20.a In your knowledge, are any members of this village involved in illegal hunting?

Yes .... ☐
No............. ☐

20.b If yes, in your opinion, why do they hunt illegally?	

Own comments:
